IUCN

Island Invasives: Eradication and Management

Proceedings of the International Conference on Island Invasives

Edited by C. R. Veitch, M. N. Clout, and D. R. Towns

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SSC Species Survival Commission

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About IUCN

IUCN, International Union for Conservation of Nature, helps the world find pragmatic solutions to our most pressing environment and development challenges.

IUCN works on biodiversity, climate change, energy, human livelihoods and greening the world economy by supporting scientific research, managing field projects all over the world, and bringing governments, NGOs, the UN and companies together to develop policy, laws and best practice.

IUCN is the world's oldest and largest global environmental organization, with more than 1000 government and NGO members and almost 11,000 volunteer experts in some 160 countries. IUCN's work is supported by over 1000 staff in 60 offices and hundreds of partners in public, NGO and private sectors around the world.

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IUCN Species Survival Commission

The Species Survival Commission (SSC) is one of six volunteer commissions of IUCN – the International Union for Conservation of Nature and Natural Resources - a union of sovereign states, government agencies and non-governmental organizations. IUCN has three basic conservation objectives: to secure the conservation of nature, and especially of biological diversity, as an essential foundation for the future; to ensure that where the earth's natural resources are used this is done in a wise, equitable and sustainable way; and to guide the development of human communities towards ways of life that are both of good quality and in enduring harmony with other components of the biosphere.

The SSC's mission is to conserve biological diversity by developing and executing programs to save, restore and wisely manage species and their habitats. A science based volunteer network comprised of nearly 7500 scientists, field researchers, government officials and conservation leaders from almost every country in the world , the SSC membership is an unmatched source of information about biological diversity and its conservation. As such, SSC members provide technical and scientific counsel for conservation projects throughout the world and serve as resources to governments, international conventions and conservation organizations. Most members are deployed in more than 100 Specialist Groups and Task Forces.

The IUCN/SSC Occasional Paper Series focuses on a variety of conservation topics including conservation overviews on a regional to taxonomic basis and proceedings of important meetings.

IUCN/SSC also publishes an Action Plan series that assesses the conservation status of species and their habitats, and specifies conservation priorities. The series is one of the world's most authoritative sources of species conservation information available to nature resource managers, conservationists and government officials around the world.

The Invasive Species Specialist Group (ISSG) - one of five Disciplinary Groups of the Species Survival Commission (SSC) of the International Union for Conservation of Nature (IUCN) - aims to reduce threats to biological diversity by increasing awareness of invasive alien species, and of ways to prevent, control and manage their spread. ISSG promotes the exchange of invasive species information across the globe and ensures the linkage between knowledge, practice and policy so that decision making is informed. Founded in 1994, the ISSG Secretariat was based at the University of Auckland in New Zealand until early 2009, when it was moved to Rome, Italy, with the appointment of the new Chair, Dr. Piero Genovesi. A Regional Pacific Office has been established in New Zealand, to serve as the Pacific node for ISSG activities and serve as the Invasive Species focal point for the IUCN Oceania Regional Office based in Fiji. ISSG is currently a network of 196 invasive species experts from over 40 countries, providing technical and scientific advice to National and Regional agencies and to civil society in developing policy and strategies to manage the risk of biological invasions. The group hosts a website (www.issg.org) and publishes a newsletter "*Aliens*"- biannually. The ISSG also hosts a listserve *Aliens-L* with more than 1085 subscribers and operates a referral service for global stakeholders. ISSG manages and maintains the Global Invasive Species Database (GISD) – recognised as a significant repository of global invasive species information. As of late 2011 the GISD featured 853 species profiles.

Centre for Biodiversity and Biosecurity

The Centre for Biodiversity and Biosecurity (www.cbb.org.nz) brings together researchers from the University of Auckland and Landcare Research, including many of New Zealand's pre-eminent experts in biosecurity, invasion ecology, conservation biology and biodiversity research. Landcare Research holds a number of nationally and internationally significant collections at Auckland – including the NZ arthropod and NZ fungi collections, the National Nematode collection and the International Collection of Micro-organisms from Plants. The University has expertise in animal behaviour, invasion ecology, plant ecology, molecular ecology, conservation biology and restoration ecology. At its Tamaki Campus it also hosts the Pacific Invasives Initiative and the Regional Pacific Office of the Invasive Species Specialist Group of SSC/IUCN. Through the CBB, the combined expertise of the University and Landcare Research provides opportunities for joint research, and a strong platform to exploit new opportunities nationally and internationally. Such interactions (including joint supervision of postgraduate students) are leading to novel research to enhance the capacity, efficiency and quality of biodiversity management, conservation and biosecurity in New Zealand and globally. The CBB is proud to be hosting this conference.

The papers and abstracts published in this book are the outcome of the conference on Island Invasives: Eradication and Management held at Tamaki Campus, University of Auckland, New Zealand, from 8 to 12 February 2010, hosted by the Centre for Biodiversity and Biosecurity (University of Auckland and Landcare Research), in collaboration with the IUCN/SSC Invasive Species Specialist Group.

This conference had "islands" and "eradication of invasive species" as the focus, with emphasis on the work done and results or learning achieved. The conference endeavoured to cover the full breadth of this work by breaking the subject down to: Gaining political, community, financial, and physical support; Eradication techniques tested and used; The immediate results of eradication operations; The longer-term outcomes as seen in the biota of the island and among communities involved; Biosecurity measures for such islands from planning to implementation.

All papers have been peer-reviewed but the content is the choice of the author. The style of presentation has been modified in consultation with the editors. Nomenclature follows international published standards.

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PREFACE

Invasive alien species: losses, gains, and engaging the public

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Recognition that invasive alien species pose a major threat to the survival of indigenous species and functioning of natural ecosystems is relatively recent (Mack *et al*. 2000). The first concerns about invaders were voiced in countries such as Australia and New Zealand after the ill-informed releases of game animals such as rabbits (*Oryctolagus cuniculus*), which then caused massive damage to agriculture. Attempts to reverse the impacts of such invasions with introduced predators simply added other invasive species to the mix and made the situation even worse (e.g., Young 2004). Often, the only solutions have been to control invasive species for short or long periods, or to remove samples of those species threatened by the invaders and hold them in safe locations. Over time, invaded ecosystems can become irreversibly changed and some, or many, indigenous species may be lost from them.

The ecological value, and potential of islands around New Zealand as conservation sites has been recognised for more than 100 years; initially by individuals such as Richard Henry (Hill and Hill 1987) and more recently by groups such as the Offshore Islands Research Group (Wright and Beaver 1986).

At the same time, islands have been used for the farming, mining, lighthouse stations, prisons, defence emplacements, and more, with these activities destroying natural ecosystems and introducing invasive alien species. There has also been deliberate introduction of edible species to islands in case of need by shipwrecked mariners. The ships wrecked on their shores often brought new invaders. Nevertheless, the natural barriers around islands offer opportunities to remove and then exclude invasive alien species. This allows regeneration and protection of ecosystems, the species in them, and possible reintroductions of species that were previously present.

Early attempts to restore natural ecosystems by removing introduced species, especially large herbivorous mammals, met with great success. This success flowed on to removal of smaller mammals, and other invasive species. By the time of the first New Zealand conference on the restoration of islands (Towns *et al*. 1990) invasive species had been removed from more than 60 islands around New Zealand, and more in other parts of the world.

The international value of this type of work was recognised in the 2001 ‘Turning the Tide’ conference held in Auckland, New Zealand (Veitch and Clout 2002), and an associated one on the science of invasive species held in Wellington.

This current volume stems from a conference held in Auckland in 2010 and attended by 240 delegates from at least 20 countries. The conference content covered any aspect of invasive species relating to natural insular ecosystems. This diverse array of subject matter is divided into four sections in the book. The first section deals with overviews and planned or attempted eradications. The second section introduces new technologies and approaches to eradications, such as dealing with multiple invasive species. Papers in the third section concentrate on the results and outcomes of eradications, especially responses by native species. The final section covers the roles and approaches that involve people, policy and invasion prevention (biosecurity).

The major purposes of holding the conference, and publishing these peer-reviewed proceedings, are to encourage and assist the management of invasive species, particularly on islands. We thank all of those who have contributed and encourage you to share and distribute this information.

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OPENING ADDRESS

Invasive species, nature's systems and human survival

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LIFE ON EARTH

This year (2010) is the United Nations International Year of Biodiversity. Theme years, even under the UN banner, can too easily pass by with little more achieved than the already committed renewing their commitment. We must not let that happen in this, the Year of Biodiversity.

Social surveys indicate that biodiversity is not a readily understood word. I do not much care for it and have been guilty of dismissing it as no more than a complicated way of saying our native plants and animals. That is wrong, of course. Biodiversity is not confined to endemic species, as it also encompasses the inter-relationship between species, the ecological health of their habitat, and the state of the ecosystem services that flow from them. It is a complex web upon which much depends. We should not shrink from the word and one of our key tasks this year is to increase general awareness and understanding of biodiversity and what it means. That includes a better realisation of our place and role as a species within nature's systems. We can do that on present knowledge, but there are significant gaps in what we know and understand about our biodiversity, and too frequently the information we have takes us no further than to advise a precautionary approach. We need to know more; much more. It is to that purpose that this conference follows on from the 2001 Island Invasives Conference held in this city. The proceedings of that 2001 meeting and the further research and collaboration that emerged from it have proved invaluable. It is timely to again meet to share knowledge and best practice, reassess priorities and set new objectives. That is the work of this conference, and the workshop that is to follow in April.

In the past, conservationists' inherent interest and intellectual thirst for greater knowledge about biodiversity was sufficient to bring us together. A belief in the intrinsic value of nature and an ethical responsibility to protect and preserve was sufficient purpose. Intrinsic value was the driving force of the legislation passed in New Zealand 22 years ago to establish the Department of Conservation, and the justification for placing one third of New Zealand's land mass, much of its fresh waters and some marine functions under conservation management.

We must maintain that high ethical commitment. It is part of what distinguishes us as a species. But to rely on it alone is to expose biodiversity to the dangers of those who do not share the same values, have the same level of appreciation, or exhibit the same degree of commitment. We have the opportunity to leverage off a growing pragmatic reasoning for protecting and enhancing our biodiversity, and there is too much at stake not to do so.

Since the 2001 conference, there has been a slow, belated and somewhat reluctant global recognition that the degradation and destruction of ecosystems on a massive scale is destroying the biodiversity that provides the services that we rely on for our prosperity, and ultimately for our survival.

This gives added purpose, and a sense of urgency, to your work. If humanity is to give itself the best chance, then we need to understand the interrelationship between species, places, and ecosystem services much better, and the critical importance of respecting, protecting, enhancing and creating biodiversity health. This situation, which we find ourselves in, is somewhat humbling. The plain simple fact is, the planet is not at risk, but we are. In its 4 to 5 billion years of existence, planet earth has been through many radical environmental changes. Species have come and gone as a result. Dinosaurs existed for 165 million years before their mass extinction in a catastrophic event. When they became extinct, new forms of life evolved in the new environmental conditions, and the planet continued to spin.

How long we, as a species, have existed depends on your evolutionary starting point, but it is certainly no more than a million years and arguably only 100,000. Either way it's considerably less than the 220 million years that New Zealand's tuatara (*Sphenodon punctatus*) have been around.

We are nothing but a brief blink of the eye in the life of the planet. It was here for billions of years before us, and if we become extinct, there is no reason to believe it will do anything other than continue on for billions of years after us. The oil peak, deforestation, climate change – none of it is of any concern to the planet. The dependency is entirely ours. If we cannot live in harmony with the natural systems that allowed our evolution and are the key to our ability to survive and thrive as a species, then the problem is ours, not the planet's.

It is of no moment to the planet whether the changes we are experiencing to our detriment are the result of our actions or natural causes. The best that the sceptics of anthropogenic climate change can do is absolve us and draw us towards threatened species status; free of blame and thus with a clear conscience. Their protestations will have no impact on nature's systems, or the inevitable outcome of degrading those systems to a point that they can no longer support us.

A HISTORY OF DEGRADATION

This situation, which we face, is neither new, nor unique. In 360 BC Plato described the Athenian's destruction of nature's systems through deforestation, and commented on their political failure to implement a solution that had been drawn up (Wright 2004). This self-destructive behaviour marked the decline of Greek supremacy. History is littered with civilizations that have sown the seeds of their own destruction by pushing nature's systems beyond their ability to sustain the society that depends on them.

This behaviour runs counter to the instinct of species to replace themselves with their finest and fittest. But is it explicable for a species with the intellectual ability to build behaviour around value systems? Environmental exploitation typically advantages the present generation while the costs lie in the future. So an ethic of self-interest is sufficient to justify capturing for yourself the immediate benefits that can accrue from environmental exploitation,

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and transfer the costs to future generations. And if your conscience is bothering you, all you need do is comfort yourself with the age-old excuse that future generations will discover new solutions to clean up your mess.

Two factors do, however, make the present-day situation significantly different to that faced by past civilizations. First, the scale of our environmental exploitation is such that the effects are borne by water and air far beyond the boundaries of the worst perpetrators. The impacts are not confined to the culprits; they are global. So no boundary smaller than the planet itself can be drawn if we are to put things right. Second, the future has caught up with us and the costs of environmental degradation that once seemed so distant as to be unreal are now ours to pay. Or if we refuse to pay, then the consequences are ours to bear.

We are not the only species that has sewn the seeds of its own demise. Scientist John Flux records that for 607 islands where the fate of introduced rabbits is known, the population died out in more than 10 percent of cases. They ate themselves out of house and home. More specifically, in 1944, 29 reindeer (*Rangifer tarandus*) were introduced to St Matthew Island, west of Alaska, by the United States Coast Guard to provide an emergency food source. The coast guard abandoned the island a few years later, leaving the reindeer. Subsequently, the reindeer population rose to about 6,000 by 1963 and then died off in the next two years to 43 animals. A scientific study attributed the population crash to the limited food supply in interaction with climatic factors. By the 1980s, the reindeer population had completely died out. The difference between us and the reindeer is that we have the intelligence to know what we are doing, see the implications, and do something to avoid it. The question is whether we have the wit to acknowledge that we cannot defeat nature, the smarts to work out what we need to do to live in harmony with it, and the will to take the necessary corrective action. The evidence to date is not comforting.

Ignorance is neither a reason, nor an excuse, for inaction. In Plato's dialogue he records how the Athenians developed a solution to the deforestation of their catchments. The problem definition and the solution were not missing, but the political will to act was. Sound familiar? Two thousand four hundred years on, the failure of the 2009 meeting on climate change in Copenhagen to address the threats posed definitively is a repeat performance.

INVASIVE SPECIES IN NEW ZEALAND

There could not be a better place to make this point than New Zealand. European colonisation took place in an era of some knowledge about the complex impacts of introduced and invasive species. But it had little impact on those who sought to recreate their home country on the other side of the world amidst a completely different native biodiversity. The results were predictable, and within short time the colonists were both engaged in trying to mitigate the impacts on their economic endeavours while continuing to introduce problem species. Don't look for the logic!

The Dog Nuisance Ordinance was passed in 1844, but its bark didn't bite. The ubiquitous Scotch thistle (*Cirsium vulgare*) was the subject of no less than five provincial government ordinances between 1854 and 1862 in attempts to prevent its spread, and various other ordinances around that time were designed to prevent gorse (*Ulex europaeus*) and broom (*Cytisus scoparius*) spreading. The weeds took no notice of the will of Parliament. The joy of seeing little bunny rabbits (*Oryctolagus cuniculus)* hopping in the fields of colonial pastures quickly wore off as they tore in to the pastoral economy and in 1876 Parliament passed the Rabbit Nuisance Act. It didn't stop the rabbits breeding like rabbits.

By 1875, introduced sparrows (*Passer domesticus*) had eaten their way through crops to a point that the farmers convinced the Canterbury Provincial superintendent that bird kill was in order. Farmers' clubs paid a bounty of a penny half-penny a dozen for sparrow eggs, and one club alone gathered in 21,000 eggs. But Cock Robin's revenge was short-lived and the sparrow plague returned.

It was 70 years before a bounty scheme was introduced to control deer numbers, with marginal effect, and despite years of debate it took 96 years for official policy to declare war on Australian brushtail possums (*Trichosurus vulpecula*). The entire effort failed dismally to turn the tide of devastation wrought by introduced pests. All the while there were, in many cases, sufficient data and warnings to have avoided the problems.

A case in point is the introduction of stoats (*Mustela erminea*). Landholders wanted to introduce stoats to control the rabbits. Ornithologists in England warned that the stoats would more likely turn on New Zealand's bird life and protests here led to Parliament passing a Bill in 1876 to prohibit their introduction. But the Upper House of the time, dominated by landowners plagued by rabbits, overruled it. The stoats came in, the rabbit problem worsened, and the bush fell silent of birdsong. Similarly John Cullen was warned against introducing heather (*Calluna vulgaris*) into Tongariro National Park but he did so anyway, motivated by a vision of a Scottish game reserve. The heather took over and remains a problem to this day, but the red grouse (*Lagopus lagopus)* that were supposed to feast on it failed to establish.

In 1872, the journal Nature editorialised against the reckless transportation of species to New Zealand and predicted: *"the importations will inevitably become the greatest of nuisances"*. One hundred and forty years on, taxpayers, ratepayers and landowners in New Zealand are forking out some $800 million a year, every year, just to control the menu of animal and weed pests that threaten our native biodiversity.

How has this happened? Stupidity, ignorance, and a selfish ethic provide some of the reasons. So does the disconnect with nature that urbanisation brings, but there is also an institutional tool that helps to drive this behaviour.

ENVIRONMENTAL DEBT

Currently, the way we describe and measure economic progress is an incentive to ignore the impacts of unsustainable natural resource use and management, and capture the benefits and subsidies from that with a clear conscience. The widely accepted international measure of an economy is gross domestic product, GDP. The International Monetary Fund is the keeper of GDP measures. It can be measured in terms of income, expenditure, or production, but over time all three produce much the same result. None of the measures take a systematic account of environmental impacts. Creating an environmental mess is good for GDP. It typically produces immediate benefit for the development at issue, and down the track the cost of cleaning up the mess generates further economic activity. This subsidisation of the developer, and transfer of costs to future generations, is built in to the system. Conventional economics discounts environmental impacts and that in turn affects the way we think, talk and act. Thus financial debt is seen as something that must be paid back. Institutionally, we reward early payment, penalise late payment and punish non payment, but we are reluctant to even talk about environmental debt and when we can't avoid it, we use the language of cost and debate whether we can afford to pay it back. We typically conclude that we can't, or certainly not in full. When the current recession revealed a collapsing financial system, some 12 trillion dollars was found in quick time to prop it up. But when nations met at Copenhagen to try and restore a collapsing environmental system, that sense of urgency and decisiveness was missing. The cupboard that stored trillions for financial collapse was apparently bare.

GDP is increasingly being questioned internationally as a suitable measure of economic growth, and not just because we look like being the generation that has to start paying back the huge cost of cleaning up the mess from previous generations. GDP measures wealth but takes little account of its distribution. If an increase in GDP translates into improved wellbeing across society, then it is a valid measure of progress. But the trend for increased wealth to be retained in fewer hands now means an increase in GDP does not necessarily translate to higher standards of living generally. Measurements show that for a number of wealthy countries, GDP is rising while general wellbeing is falling. That is a recipe for social instability, and social instability is dangerous.

If GDP is failing as a measure of both social stability and environmental sustainability then surely that is a powerful incentive to find a new construct that measures true progress. It is no easy task to construct one. The simple solution is to balance economic, social and environmental considerations and reach a pragmatic compromise. But that won't do it. Living in harmony with nature's systems; living sustainably, is not apart from the economy, it is a key component of it. Nature's systems lie at the base of any economy. If they are not functioning efficiently, then the economy cannot function efficiently. If we destroy them, we destroy the economy. Accepting a definition and measure of wealth that discounts the impact of our activity on those systems ultimately acts against our own interests. It exposes us to the risk of threatened species status, and ultimately to extinction as a species.

THE VALUE OF SPECIES AND ECOSYSTEMS

I make no apology for spending this time on economic measures at a conference on invasive species. It lies at the heart of why the loss of habitat, and the accompanying loss of species, is so poorly appreciated and accounted for in public policy.

The context it creates for you is to appreciate the need for conservationists not to appeal to intrinsic value alone in the battle to save our species. We must be able to argue their importance in the natural cycles and systems that humanity relies on to survive and prosper. The health of our native species indicates the health of our ecosystems, which in turn determine the health of the services that flow from them, and upon which we rely. We are dependent on this natural capital. That is the economics of ecosystems and biodiversity. Investing in it provides a healthy return.

Since the last conference there has been good progress in controlling invasive species on both the prevention and control fronts. But the declining state of our biodiversity requires even more rapid progress. If we are to be more effective in this critical work, and we need to be, then you are the people who are going to provide the knowledge to make that happen. This is a great opportunity to share your thinking and determine what needs to be done in the decade ahead. I wish you every success in this endeavour.

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KEYNOTE ADDRESS

Are we turning the tide? Eradications in times of crisis: how the global community is responding to biological invasions

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Abstract Biological invasions are a major driver of the ongoing loss of biodiversity, and if the global community wants to reverse this trend it is crucial that formal commitments be transformed into action. On the basis of the more than 1000 eradications attempted worldwide, we can now say that eradication projects are a powerful conservation tool that has contributed to improving the conservation status of several threatened species. The growing sophistication of the scientific and technical basis of eradications now allows us to target much larger areas than in the past, and the eradication of species in much more challenging taxonomic groups. Also, it is now possible to minimise the risk of undesired effects of eradications, ensuring selectivity of the removal methods and minimised impacts on the environment. Despite these advances, the implementation of removal campaigns is still limited, partly by prejudices and ignorance, but also by serious concerns from a part of society, which we need to take seriously into account. It is important to ensure a correct and transparent flow of information. If the global community wants to fulfil the commitment to reverse the present rate of biodiversity loss, it is crucial to increase the application of invasive species removal campaigns and to support large scale flagship projects, as well as by developing frameworks that permit the rapid detection and removal of newly established invasive species.

Keywords: Overview, CBD, invasive alien species, management

INTRODUCTION

In 2002, the global community committed to achieve, by 2010, a significant reduction in the loss of biodiversity and - in order to verify what has been done to reach this goal - the UN declared 2010 the International Year of Biodiversity. Unfortunately, the indicators that have been collected in recent years show that there is little to celebrate. The global rate of biodiversity loss appears to have increased, and so have most of the pressures affecting the diversity of life on earth. For example, the overall status of birds in different regions of the world from 1988 to 2008 has declined, with the proportion of threatened birds increasing from 11.1% to 12.2% in that 20-year period (Butchart *et al*. 2004), and other taxa appear to be in a worse conservation shape (Vie *et al*. 2009). The continuing loss of biodiversity is particularly alarming on island ecosystems, which host a large proportion of the world endemics. Most threatened species are, in fact, found on islands (Vié *et al*. 2008); about one-fifth of the world's threatened amphibian fauna, one-quarter of the world's threatened mammals and more than one-third of the world's threatened birds are endemic to island biodiversity hotspots (Fonseca *et al*. 2006). And it is these hotspots that have had most of the recent extinctions; 88% of known bird extinctions have been on islands (Butchard *et al*. 2006), mostly because of biological invasions. Invasive species have in fact been identified as a key factor in 54% of all known extinctions, and the only factor in 20% of extinctions (Clavero and Garcia-Berthou 2005).

ARE WE TURNING THE TIDE?

Instead of recording a mitigation of the drivers of biodiversity loss, all the evidence confirms that the number of invasive alien species is rapidly growing in all environments and among all taxonomic groups (Genovesi *et al*. in press), raising extinction risks for birds, mammals and amphibians (Clavero *et al*. 2009). The most effective way to address this threat is through a combination of prevention measures, early detection at and near borders, prompt eradication of newly-arrived unwanted aliens, and effective management of established invasive species. Eradication is thus a key component of a global response to invasions, and for this reason Dan Simberloff, in his opening speech at the 2001 international conference on island invasive species, stressed the urgent need for a much wider application of this conservation tool. He challenged decision makers and practitioners to be much more ambitious in their efforts to combat invasions, overcoming the prejudices and groundless opposition that have so far limited the potential range of application of removal campaigns. In the present contribution I will thus discuss developments since 2001, and try to assess to what extent we have been able to respond to the call for more action that was launched on that occasion.

ERADICATION: AN EFFECTIVE RESPONSE TO INVASIONS

There is increasing evidence that successful invasive species removal campaigns have played a crucial role in improving the conservation status of several taxonomic groups. Many endemic and rare species have recovered following the eradication of invasives threatening their persistence. An assessment of red list data has shown that 11 bird, five mammal and one amphibian species have improved their conservation status as a result of eradications of invasive species (McGeoch *et al*. 2010). These positive outcomes are also the result of the significant improvements in the science of eradication over recent decades. As discussed by Alan Saunders in this volume (Saunders *et al*. 2011), the number of multi-species eradications is constantly increasing, and the experience gathered in the last 20 years now minimises the risk of undesired effects of eradications, ensuring selectivity of the removal methods and minimised impacts on the environment. Furthermore, we are increasing our ability to predict potential ecosystem changes caused by the removal of invasive species, and adaptive implementation of eradications has prevented or rapidly mitigated potential unexpected chain reactions (see Courchamp *et al*. 2011; Morrison 2011). We can now target much more challenging taxonomic groups, such as plants and terrestrial invertebrates. Regarding the latter, up to a few years ago invertebrates were generally considered as not eradicable, with few exceptions. In his paper of 2002, Simberloff stressed the need to test whether eradication of insects on continents was really out of the question. The

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general pessimism challenged by Simberloff resulted from several unsuccessful eradication campaigns, such as the attempt to remove the red imported fire ant (*Solenopsis invicta*), from the US. However, it must be stressed that these attempts have significantly increased the technical basis of eradication, recently allowing several successful eradications: for example in the Galapagos, but also in several mainland areas of Australia and New Zealand (Hoffman *et al*. 2010; Hoffmann 2010).

GLOBAL OVERVIEW OF ERADICATIONS

Several recent reviews of eradications have been published (Veitch and Clout 2002; Nogales at al. 2004; Campbell and Donlan 2005; Genovesi 2005; Howald *et al*. 2007; Genovesi and Carnevali 2011), with the most up-to-date and comprehensive one for vertebrate eradications on islands being in this volume (Keitt *et al.* 2011).

These publications, and the data presented at the 2010 conference, show that globally 1129 eradication programmes have targeted alien species of plants or animals on the mainland or islands. This number is very likely an underestimate, since many eradications go unreported, especially those of plants. Of the projects I considered, 86% were reported as successful (n=911; 819 vs. 93), and 97.07% were carried out on islands (n= 1,129; 1096 vs. 33). Some 94.6% of reported eradications targeted vertebrates (n= 1,119; 1059 vs. 60), but as already mentioned, this in part reflects the difficulty of accessing plant management data, as well as records of invertebrates eradications (i.e. no global review of mosquito eradications has been published so far, to my knowledge).

Eradications range from large scale programes addressing widely distributed invasives to the removal of a few individuals established in a still restricted range. Both extremes are of crucial importance. We need large scale, ambitious programmes to verify the potential of eradications, and at the same time to show the public and decision makers the results that can be obtained. At the same time we need examples of routine detection and localised eradication projects, to show how invasions can be addressed at their very early stages, through well-designed and well-implemented operational frameworks.

Regarding the first case, several programmes that have been initiated and, if successfully completed, will indeed provide excellent evidence on the potential of this tool. One example is the ongoing eradication of the ruddy duck (*Oxyura jamaicensis*) from Europe. This programme is particularly challenging. The species was imported intentionally into the UK where it became established in the 1960s. The ruddy duck hybridises with the endangered white-headed duck (*O. leucocephala*) (Muñoz-Fuentes *et al*. 2007), putting at risk the survival of this rare species, which has a total population of only 3000 pairs in the entire Palaearctic (Henderson and Robertson 2007). Removing the introduced species is particularly complex for several reasons. Firstly, the core European population of the ruddy duck is in UK, and it is thus in this country that most of the control actions have to be undertaken. However, reproduction is mostly in Spain, and so no crucial impact is recorded in the country that is responsible for the main removal operations. Furthermore, individuals or small populations of ruddy ducks occasionally appear in other European countries, such as France, the Netherlands or the Baltic countries. If any of these countries will not enforce the needed management activities the entire eradication programme may be undermined. But despite these complex challenges, the results of European action so far appear very encouraging. A Pan-European action plan was approved by parties of the Bern Convention in 1999. The eradication of the UK population of ruddy duck commenced in 2005 by the competent authorities (see Henderson, 2009 for an update). The eradication cost of the campaign (£3.3 M for the first phase of work) has partly been covered with financial support from the European Commission. As a result, 90% of the UK population had been removed by winter 2008/2009. Despite some opposition from animal rights groups, the control programme had the support of all major British conservation organisations and most of the general public. Hybrids are systematically culled in Spain, by a removal protocol that minimises the risk of removing the native species. As a result of these control activities, the Spanish population of white-headed ducks has grown from the 22 breeding pairs in 1977 to the present 2100-2600. When the eradication is completed, this will indeed represent a unique example of cooperation for conservation, and of the results that can be obtained with adequate planning and effective international coordination frameworks.

Another example of encouraging international cooperation to carry out an eradication is the planned removal of the Canada beaver (*Castor canadensis*) from Tierra del Fuego (Malmierca *et al*. 2011). The beaver was introduced to Tierra del Fuego in 1946 for fur production and has established in over 27,000 km of waterways and 7,000,000 ha of Argentina and Chile. This species has a huge impact on forests, steppes and meadows, as well as on infrastructure; calling for the launch of a coordinated eradication campaign. However, cooperation between Chile and Argentina was inhibited by the tensions and conflicts that have characterised the relations between the two countries after the Beagle Conflict in the 1970s and 1980s. Despite these diplomatic tensions, in 2006 Chile and Argentina signed a cooperation agreement for eradicating the beaver. A feasibility study completed in 2008 by an international team, concluded that the eradication is possible although very difficult, and will require at least 9 years work, and an overall investment of at least 33 million USD.

But even if these large scale projects provide good examples of what can be achieved with adequate commitment and resources, it is also crucial that countries improve their ability to carry on prompt eradications immediately after a new invader has arrived into their territory. Prompt detection and response is, in fact, by far the most effective and economically convenient way to address new invaders, as shown by a review of plant eradications carried out in New Zealand by Harris and Timmins (2009). They found that early removal of plants costs on average 40 times less than removals carried out after an invasive plant has widely established.

One example of an effective approach to early detection and rapid response to invasions is the California Weed Action Plan (Schoenig 2005). This plan is enforced through a partnership between state agencies and key stakeholders. It is based on an official list of noxious weeds for which it is mandatory to act promptly, and is based on a budget of about USD 2.5 M/yr. Early detection of new infestations is ensured by the involvement of a network of biologists, and trained farmers and volunteers. The State weed programme provides grants for local weed control activities of about USD 1.5 M/yr. The application of the action plan has led to the successful removal of over 2000 infestations, and to the complete eradication of 17 weeds from the State.

CONCLUSIONS

Biological invasions are growing at an alarming rate and are a major driver of biodiversity loss, but also affect our economy, health, and the ecosystem services we rely on. The most effective way to reduce these threats is to enforce prevention measures, by establishing stringent biosecurity policies at the national, regional and global scale. However, when prevention fails, eradication is indeed one of the most concrete and cost-effective responses to invasions, and this tool can eventually reverse the present rate of biodiversity loss. The more than 1000 recorded eradications have reflected significant technical advances that now allow the targeting of much more challenging species and areas than in the past, and allow minimal undesired environmental effects. For example, we now know that - with adequate planning, effective techniques and sufficient resources – many ant infestations could be removed from the world. And projects such as the ongoing eradication of the ruddy duck in Europe indicate that many widely established invasive species – such as the red fox (*Vulpes vulpes*) in Tasmania (Parkes and Anderson 2011) or the beaver in Tierra del Fuego - could be removed with long-term commitment and adequate resources.

However, in most cases eradications are still realised at the single small-island level, there are no examples of completed large scale flagship projects – carried on invasive species widely established on mainland - and there are very few cases of structured national frameworks ensuring early detection and rapid removal of new invasions, as in the case with the Californian weed programme. The still very limited implementation of eradication programmes is in part the result of the opposition and prejudices of different sectors of the society. For example, fierce opposition by a few animal rights groups contributed to the failure of the attempt to eradicate the American grey squirrel (*Sciurus carolinensis*) from Northern Italy (Genovesi and Bertolino 2001; Bertolino and Genovesi 2003). And the growing opposition in New Zealand to the use of aerial baiting (expressed for example in the film "*1080: Poisoning Paradise*") or petitions to stop the control of feral camel (*Camelus dromedarius*) populations in Australia, are other more recent examples of this phenomenon.

The opposition to eradications also finds support in the lack of real commitment by countries. In fact, although 55% of countries have implemented specific national legislation, and many more have formally committed to increase their efforts to tackle the threat of invasions (McGeoch *et al*. 2010), the level of on-the-ground action has not grown apace with these largely token formal commitments. A more structured application of eradications will require effective national policies, clear legal tools, and financial and institutional support. Apart from existing obstacles at the national level, action on a global scale is also far from being satisfactory. The Convention on Biological Diversity in 2002 led the conference of the parties to adopt the decision VI/23 on invasive alien species, and provided detailed guiding principles for its implementation. However, no global binding tool on invasions has been adopted, and the guiding principles have thus remained largely on paper. This lack of global action was stressed by the G8 Environment meeting held in 2009 in Syracuse, which adopted a final charter on biodiversity that included the urgent need to develop global early warning and rapid response systems.

If the global community really intends to reverse the present trends of biodiversity loss, it is urgent that world leaders translate all the technical work done in the last 30 years, as well as turning the adopted commitments into concrete action, particularly by giving priority to addressing biological invasions on islands, as this may significantly curtail the continuing decrease of species numbers.

The scientific community must communicate better the value of eradications, building on the many success stories; "flagship" large-scale projects are crucial in this respect, and it is important to support these campaigns. We must also address the growing concerns in some sectors of society (see Cowan and Warburton 2011), reducing as much as possible the undesired side effects of eradications, and always ensuring a transparent flow of information. Last but not least, the scientific community should encourage the development of a global programme of work based on an agreed set of priorities and with effective early warning systems. This is a crucial condition for ensuring rapid responses to new invasions.

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PRESENTED PAPERS

Overviews and Proposals

Selected summaries of island alien species invasions and eradication work to date; and plans, preparation work and considerations for the future.

A pilot study for the proposed eradication of feral cats on Dirk Hartog Island, Western Australia

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Abstract Feral cat eradication is planned for Dirk Hartog Island (620 km^2), which is the largest island off the Western Australian coast. The island, in the Shark Bay World Heritage Property, once supported at least 13 species of native mammals but only three species remain. Since the 1860s, Dirk Hartog Island has been managed as a pastoral lease grazed by sheep and goats. Cats were probably introduced by early pastoralists and became feral during the late 19th century. Dirk Hartog Island was established as a National Park in November 2009, which provides the opportunity to eradicate feral cats and reconstruct the native mammal fauna. A 250 km^2 pilot study was conducted on the island to assess the efficacy of aerial baiting as the primary technique for the eradication campaign. Initially, cats were trapped and fitted with GPS data-logger radio-collars. The collars were to provide information on daily activity patterns, to determine detection probabilities, and to optimise the proposed spacing of aerial baiting transects and the monitoring track network for the eradication. Baiting efficiency was determined from the percentage of radio-collared cats found dead following the distribution of baits. Fifteen cats were fitted with radio-collars and 12 (80%) of the cats consumed a toxic bait. Pre- and post-baiting surveys of cat activity were also conducted to record indices of activity at sand plots and along continuous track transects. Significant reductions in these indices after baiting coincided with declines of the same magnitude as radio-collar returns. Information collected in this pilot study should help to improve kill rates and has increased confidence that eradication can be successfully achieved.

Keywords: *Felis catus*, baiting, trapping, GPS collars, telemetry

INTRODUCTION

There is extensive evidence that domestic cats (*Felis catus*) introduced to offshore and oceanic islands around the world have had deleterious impacts on endemic land vertebrates and breeding bird populations (eg. van Aarde 1980; Moors and Atkinson 1984; King 1985; Veitch 1985; Bloomer and Bester 1992; Bester *et al*. 2002; Keitt *et al*. 2002; Pontier *et al*. 2002; Blackburn *et al*. 2004; Martinez-Gomez and Jacobsen 2004; Nogales *et al*. 2004). Insular faunas that have evolved for long periods in the absence of predators are particularly susceptible to cat predation (Dickman 1992).

On Dirk Hartog Island (620 km^2), which is the largest island off the Western Australian coast (Abbott and Burbidge 1995), 10 of the 13 species of native terrestrial mammals once present are now locally extinct (Baynes 1990; McKenzie *et al.* 2000) probably due to predation by cats (Burbidge 2001; Burbidge and Manly 2002). The extirpated species of mainly medium-sized mammals include: boodie (*Bettongia lesueur*), woylie (*Bettongia penicillata*), western barred bandicoot (*Perameles bougainville*), chuditch (*Dasyurus geoffroii*), mulgara (*Dasycercus cristicauda*), dibbler (*Parantechinus apicalis*), greater stick-nest rat (*Leporillus conditor*), desert mouse (*Pseudomys desertor*), Shark Bay mouse (*Pseudomys fieldi*), and heath mouse (*Pseudomys shortridgei*). Only smaller species still inhabit the island: ash-grey mouse (*Pseudomys albocinereus*), sandy inland mouse (*Pseudomys hermannsburgensis*), and the little long-tailed dunnart (*Sminthopsis dolichura*). It is possible that the banded hare-wallaby (*Lagostrophus fasciatus*) and rufous hare-wallaby (*Lagorchestes hirsutus*) were also on the island as they are both on nearby Bernier and Dorre Islands, and were once on the adjacent mainland. The island also contains threatened bird species including: Dirk Hartog Island white-winged fairy wren (*Malurus leucopterus leucopterus*), Dirk Hartog Island southern emu-wren (*Stipiturus malachurus hartogi*), and the Dirk Hartog Island rufous fieldwren (*Calamanthus campestris hartogi*). A population of the western spiny-tailed skink (*Egernia stokesii badia*) found on the island is also listed as threatened.

Since the 1860s, Dirk Hartog Island has been managed as a pastoral lease grazed by sheep (*Ovis aries*) and goats (*Capra hircus*). More recently, tourism has been the main commercial activity on the island. Cats were probably introduced by early pastoralists and became feral during the late 19th century (Burbidge 2001). The island was established as a National Park in November 2009, which now provides the opportunity to reconstruct the native mammal fauna. Dirk Hartog Island could potentially support one of the most diverse mammal assemblages in Australia and contribute significantly to the long-term conservation of several threatened species. Successful eradication of feral cats would be a necessary precursor to any mammal reintroductions.

Baiting is the most effective method for controlling feral cats (Short *et al*. 1997; EA. 1999; Algar *et al*. 2002; Algar and Burrows 2004; Algar *et al*. 2007; Algar and Brazell 2008) when there is no risk posed to non-target species. A 250 km^2 pilot study was conducted on the island in March-May 2009 to assess the efficacy of aerial baiting, the primary technique to be used in the proposed eradication campaign.

Prior to the baiting programme, cats were fitted with GPS data-logger radio-collars to provide detailed information on cat activity patterns. These data will be used to plan the spacing of flight transects so that feral cats have the greatest chance of encountering baits within the shortest possible time. The goal is to provide the most cost-effective baiting regime.

Feral cat activity at plots along survey transects, usually along existing tracks, can be used before and after baits are spread to determine the impact of baiting programmes. Where eradication of feral cats is intended, such as on islands, such surveys are often used to locate cats that have survived the baiting programme. However, these surveys are often conducted along cross-country transects as track networks are usually limited. Rapid detection of cats surviving the initial application of baits is crucial if survivors are to be eradicated before they can reproduce.

Information on movement patterns can be used to assess rates of encounter (detection probabilities) for survey transects at various spacings across the island. It will then be possible to select the best spacing for these transects to optimise encounter frequency during surveys.

In conjunction with this study, other aspects of feral cat control that were investigated included the potential use of the toxicant PAPP (para-aminopropiophenone) in a 'Hard Shell Delivery Vehicle' (see Johnston *et al*. 2010; Johnston *et al*. 2011) and facets of movement patterns and home range use that will be reported elsewhere (e.g., Hilmer 2010, Hilmer *et al.* 2010). This paper focuses on the results of the baiting programme and how the information will be used to improve eradication efficacy.

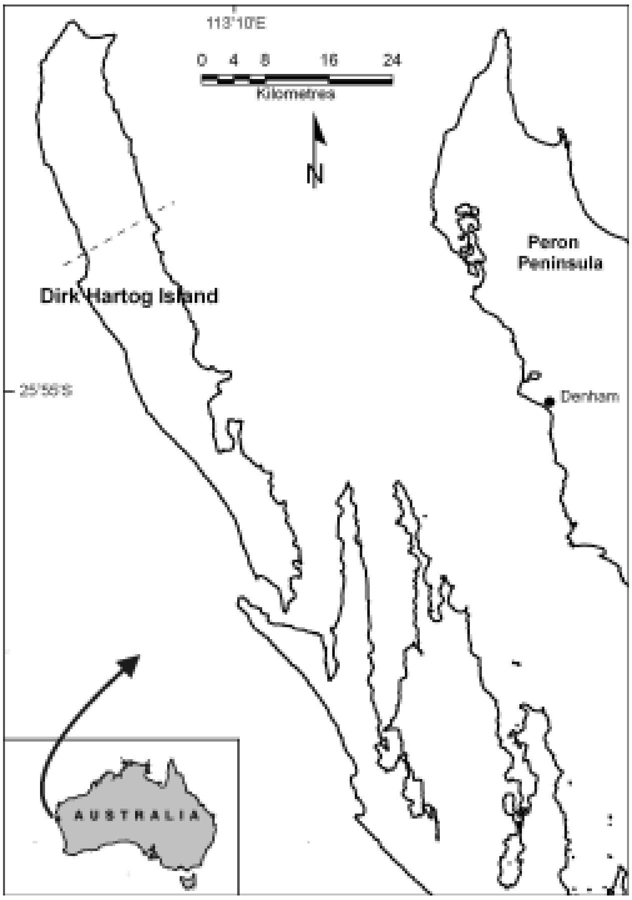

Fig. 1 Study area on Dirk Hartog Island; dashed line represents southern boundary of study area.

MATERIALS AND METHODS

Study site

Dirk Hartog Island (25°50'S 113°0.5'E) within the Shark Bay World Heritage Property, Western Australia (Fig. 1) is approximately 79 km long and a maximum of 11 km wide. This study was conducted over a 250 km^2 area at the north of the island using the track between Sandy Point and Quoin Head as the southern boundary (Fig. 1).

Vegetation on the island is generally sparse, low and open and comprises spinifex (*Triodia*) hummock grassland with an overstorey of *Acacia coriacea*, *Pittosporum phylliraeoides* over *Acacia ligulata*, *Diplolaena dampieri*, *Exocarpus sparteus* shrubs over *Triodia* sp., *Acanthocarpus preissii* and *Atriplex bunburyana* hummock grasses, chenopods or shrubs (Beard 1976). Adjacent to the western coastline is mixed open chenopod shrubland of *Atriplex* sp., *Olearia oxillaris* and *Frankenia* sp. and slightly inland in more protected sites, *Triodia plurinervata*, *Triodia* sp., *Melaleuca huegelii*, *Thryptomene baeckeacea* and *Atriplex* sp.. There are patches of bare sand and a few birridas (salt pans). On the east coast there are patches of mixed open heath of *Diplolaena dampieri*, *Myoporum* sp. and *Conostylis* sp. shrubs (Beard 1976).

The climate of the region is 'semi-desert Mediterranean' (Beard 1976; Payne *et al*. 1987). Mean maximum daily temperatures are up to 38° C. in summer and can decline to 21° C. during winter. January and February are the hottest months while June and July are the coolest. Rainfall averages 220 mm per year, mostly from May-July (Bureau of Meteorology).

Cat trapping and radio-collaring

Feral cats were trapped on the track network in padded leg-hold traps, Victor 'Soft Catch' traps No. 3 (Woodstream Corp., Lititz, Pa.; U.S.A.) using a mixture of cat faeces and urine as attractant. Trapped cats were sedated with an intramuscular injection of 4 mg/kg Zoletil 100 (Virbac, Milperra; Australia). Sex and body weight were recorded and a GPS data-logger/radio-telemetry collar with mortality signal (Sirtrack Ltd, New Zealand) was fitted. The weight of the collar (105 g) restricted their use to cats weighing >2.1 kg (5% of bodyweight). The collars were factory programmed to take a location fix every 10 (n=8), 40 (n=2) and 80 (n=6) minutes. Differences in location fix times were due to variation in other study requirements. Cats were released at the site of capture.

Eradicat baits were delivered by air three weeks after cats were released. Collars were retrieved after individual cats died as indicated during daily monitoring using VHF telemetry. Data downloaded from GPS-collars with equipment provided by the manufacturer included: date, time, latitude and longitude, number of satellites and horizontal dilution of precision (HDOP). The HDOP is the likely precision of the location as determined by the satellite geometry, which ranges from 1-100 (Sirtrack GPS Receiver Manual).

Simulation modelling of cat detectability

Analysis was performed in R2.9.0. (R Development Core Team 2009). Data from all cats alive immediately prior to baiting were utilised but only locations with an HDOP < 6 were used for the analysis. HDOP values between 6-10 are less precise (e.g., Moseby *et al*. 2009) and are more likely to have shown cats crossing transect lines not actually crossed. For each simulation, four sets of transect lines were located at random starting points and spaced at intervals of 500, 1000, 1500 and 2000 m respectively. Transect lines ran parallel to the long axis of the island and the orientation of the dune system. This was the preferred course for survey transects for logistic reasons and to minimise disturbance and erosion to dunes by the All Terrain Vehicles (ATVs) to be used during the monitoring. For each set of transect lines, the time from initial collaring of each cat to when it would have first crossed the transect line was determined. This process was repeated 5000 times with different random starting locations for the transect lines each time. For each transect line spacing, the 95th percentile of the time to cross a transect for each cat was interpreted as the time required to be 95% sure of detecting that cat during transect surveys.

Surveys of cat activity

Two independent methods were used to monitor baiting efficacy: 1) the percentage of radio-collared cats found dead after the baiting programme; 2) surveys of cat activity

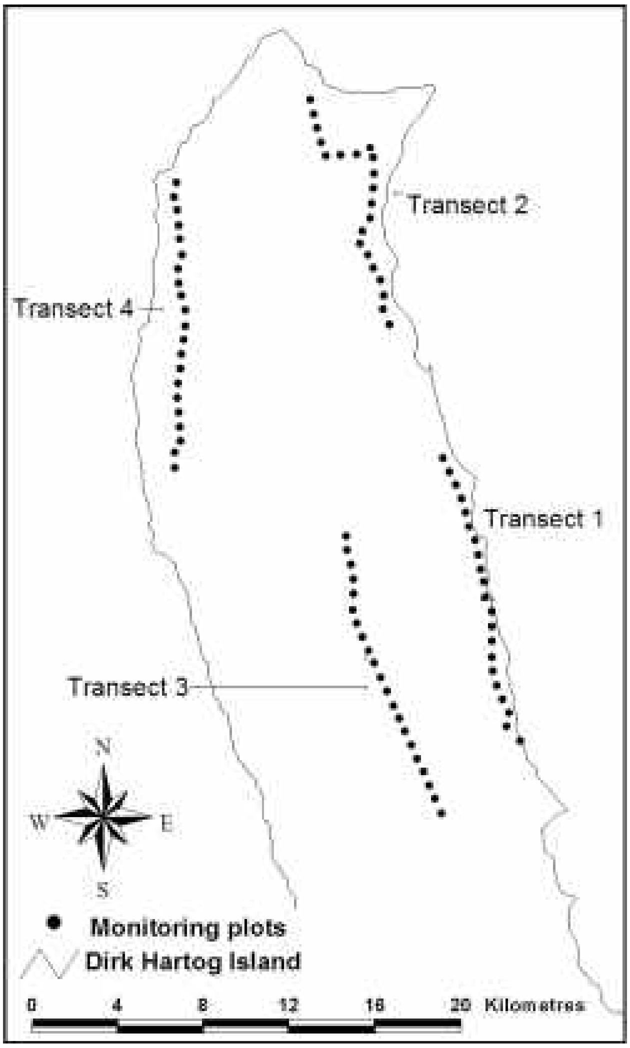

Fig. 2 Location of transects (T1-4) and monitoring plots.

at sand plots and along continuous track transects to derive indices of activity. Differences in the indices obtained pre- and post-baiting were then used as a measure of baiting efficacy.

Four track survey transects (i.e. spatial replicates) were established across the site. Each transect was 10 km in length along existing tracks and these provided a broad coverage of the entire area. Transects were separated by approximately 5 km to ensure independence. Twenty marked sand plots, positioned across the width of the tracks and located at 0.5 km intervals along each transect, were used to survey feral cat activity (Fig. 2). An audio lure (Felid Attracting Phonic, Westcare Industries, Western Australia) and an olfactory lure (Cat-astrophic, Outfoxed, Victoria) were used to attract cats to the sand plots during the two survey periods. The audio lure was concealed within a bush at the rear of the sand plot and the olfactory lure was presented on cotton wool tied to the vegetation. Both lures were removed outside the survey periods. Vehicle traffic and the limited access to the south of the study area precluded monitoring similar transects in a control (non-baited) site.

Each plot was observed for the presence or absence of tracks, as it was not possible to determine the number of intrusions by individual animals onto the plot. The plots were swept daily to clear evidence of previous activity. Cat activity at the sand plots was recorded over five consecutive nights during two survey periods to generate a Plot Activity Index (PAI). This index is expressed as the mean number of sand plots visited by the target species per night. The PAI is formed by calculating an overall mean from the daily means (Engeman *et al*. 1998; Engeman 2005). The VARCOMP procedure within the SAS statistical software package produced the variance component estimates.

The survey tracks had a sandy surface substrate that also enabled the use of a continuous 'Track Count Index' (TCI) to monitor daily activity along the length of the four transects. Imprints of individual animals were differentiated on the basis of location on the transect. An imprint was assigned to an individual animal if no other imprint was present on at least the previous 1 km of transect. Subsequent imprints were also assigned to that individual unless at least 1 km was traversed with no new imprints present, or the imprint could be clearly differentiated on the basis of size or the direction of travel or the direction of entry/exit to and from the transect. Each time new cat tracks were encountered along the transect, information was recorded on the direction of movement (i.e. whether the animal walked along the track or crossed it), distance of the tracks from the start of the transect, and whether more than one animal was present. Data were also noted on the distance that the tracks remained on the transect. Track counts were conducted from ATVs driven at a speed of <10 km h^{-1}. Transects were swept on the return journey using a section of heavy conveyer rubber and chains towed behind the ATV. The total number of cats was summed over the sampling days for each transect and the TCI was the transect mean expressed as the number of cats/10 km of transect.

Comparison of the indices pre- and post- baiting were analysed using a 'z'-test (for sample sizes greater than 30 i.e. PAI data) or the 't'- test (for samples less than 30, i.e. TCI data) (Elzinga *et al*. 2001).

Baits and baiting programme

The feral cat bait (*Eradicat*) (see detailed description in Algar and Burrows 2004; Algar *et al*. 2007) can effectively reduce cat numbers (Algar *et al*. 2002; Algar and Burrows 2004; Algar *et al*. 2007).The baits contain 4.5 mg of directly injected toxin '1080' (sodium monofluoroacetate). In addition, 3600 baits were manually implanted with a Rhodamine B 'Hard Shell Delivery Vehicle' (HSDV) made available as a part of a separate study (Johnston *et al.* 2010; Johnston *et al*. 2011). Rhodamine B is an efficient systemic biomarker for determining bait consumption by feral cats and a wide range of non-target species (Fisher 1998). When Rhodamine B is consumed, the compound causes short-term staining of body tissues, digestive and faecal material with which it comes in contact.

To optimise efficacy, the baiting campaign needed to be conducted before late autumn/winter rainfall began in May (long term Bureau of Meteorology data). On 19 April, a dedicated baiting aircraft dropped the baits at previously designated bait drop points (Johnston *et al*. 2010). The baiting aircraft flew at a nominal speed of 130 kt and 500 ft (Above Ground Level) and a GPS point was recorded on the flight plan each time bait left the aircraft. The bombardier released a bag of 50 baits into each 1 km map grid, along flight transects 1 km apart, to achieve an application rate of 50 baits km^{-2} (Fig. 3). Baits containing the Rhodamine B HSDVs were strategically dropped into the map grids immediately surrounding the locations of the collared cats. All other areas were baited with conventional *Eradicat* baits (i.e. without the Rhodamine B HSDV). The ground spread of 50 baits is approximately 250 x 150 m (D. Algar unpub. data). The Western Australian guidelines for use of 1080 baits provides for 'Bait Exclusion Zones'

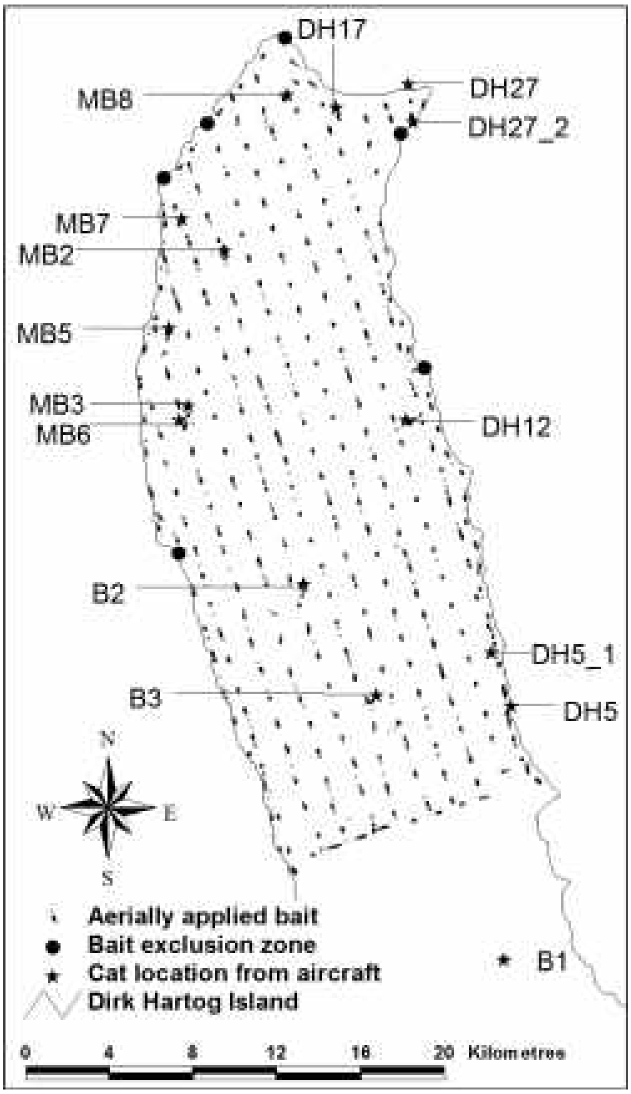

Fig. 3 Pattern of bait distribution, and locations of collared feral cats on the day of baiting. Note that cat B1, outside the baited zone, was found dead before the baiting, cause of death is unknown. Cat DH29 was not located from aircraft, and therefore it is missing from the map.

of 500 m radius at and around sites subject to high human visitation. No baits were applied to seven such sites within the study area (Fig. 3).

Immediately before baiting, locations were plotted from fixes obtained from an aircraft for all but one of the collared cats (Fig. 3). Daily monitoring of radio-collared cats was undertaken from 21 April, using either a handheld yagi antenna or a vehicle mounted omni-direction antenna connected to a VHF receiver. The death of cats was indicated by a change in pulse rate from the collars, as it switched to mortality mode following 24 hours of no movement.

Additional *Eradicat* baits were laid by hand in the vicinity of all collared cats that were still alive on 27 April. Cats surviving till 1 May were located using radio telemetry and shot to recover the GPS data-logger collars.

RESULTS

Cat trapping and radio-collaring

Twenty-one cats were trapped, comprising 13 males and eight females (Table 1). Seventeen cats were trapped on the coastal tracks and four along the central track, where cat activity appeared lower. Bodyweight (mean ± s.e.) for males was 4.3 ± 0.3 kg and 2.9 ± 0.3 kg for females. Sixteen radio-collars were available; five cats were released without a collar, four of which were under the established collar to body mass ratio (> 2.1 kg). All cats were in excellent body condition, with large deposits of body fat.

A compilation of all location data obtained from the data-logger collars is presented in Fig. 4. Analysis of daily movement patterns, pooled for all cats, indicates that the time (mean ± s.e.) to encounter transect lines spaced at 500, 1000, 1500 and 2000 m was 1.0 ± 0.2, 1.8 ± 0.5, 4.6 ± 1.1 and 12.2 ± 3.2 days respectively. The time to cross

Table 1 Morphological details and GPS data-logger collar activity period for feral cats trapped on Dirk Hartog Island.

Cat No.	Sex (M/F)	Weight (kg)	Data-logger activity period (GPS sampling frequency)
B1	M	3.8	25 March – 19 April (80 mins)
B2	F	3.5	29 March – 18 April (10mins)
B3	F	3.7	29 March – 24 April (10 mins)
B5	M	1.5	Not collared
DH5	M	5.1	28 March – 20 April (10 mins)
DH5_1	M	4.2	28March – 16 April (10 mins)
DH8	F	2.0	Not collared
DH12	M	5.0	28 March – 15 April (10 mins)
DH17	M	5.0	28 March – 23 April (10 mins)
DH26	F	2.0	Not collared
DH27	M	5.1	30 March – 8 May (40 mins)
DH27_2	M	4.5	31 March – 21 April (40 mins)
DH29	M	4.7	30 March – 7 May (80 mins)
MB1	F	1.8	Not collared
MB2	M	3.2	29 March – 22 April (80 mins)
MB3	M	3.2	25 March – 22 April (80 mins)
MB4	F	2.7	Not collared
MB5	F	3.7	28 March – 10 April (10 mins)
MB6	M	4.7	28 March – 18 April (80 mins)
MB7	F	3.5	29 March – 21 April (80 mins)
MB8	M	5.5	27 March – 8 April (10 mins)

Table 2 The time to encounter transect lines spaced at 500, 1000, 1500 and 2000 m for individual cats.

Cat ID	Number of days to be 95% sure of detecting cat at transect spacings			
	500 m	1000 m	1500 m	2000 m
B2	0.5	1.5	6.5	8.6
B3	1.0	1.0	1.7	10.6
DH5	0.4	0.4	12.6	14.8
DH5_1	0.5	0.6	8.5	11.5
DH12	0.5	0.6	0.6	0.7
DH17	0.9	5.5	5.9	9.7
DH27	1.5	1.5	1.7	2.4
DH27_2	0.1	1.0	3.7	3.8
DH29	2.5	6.5	6.6	12.6
MB2	0.6	0.6	1.5	5.6
MB3	0.9	1.9	1.9	18.5
MB5	0.5	0.6	0.8	>40
MB6	1.5	1.6	13.0	>40
MB7	2.8	3.5	3.5	3.7
MB8	0.5	0.7	0.7	0.8
mean ± s.e.	1.0 ± 0.2	1.8 ± 0.5	4.6 ± 1.1	12.2 ± 3.2

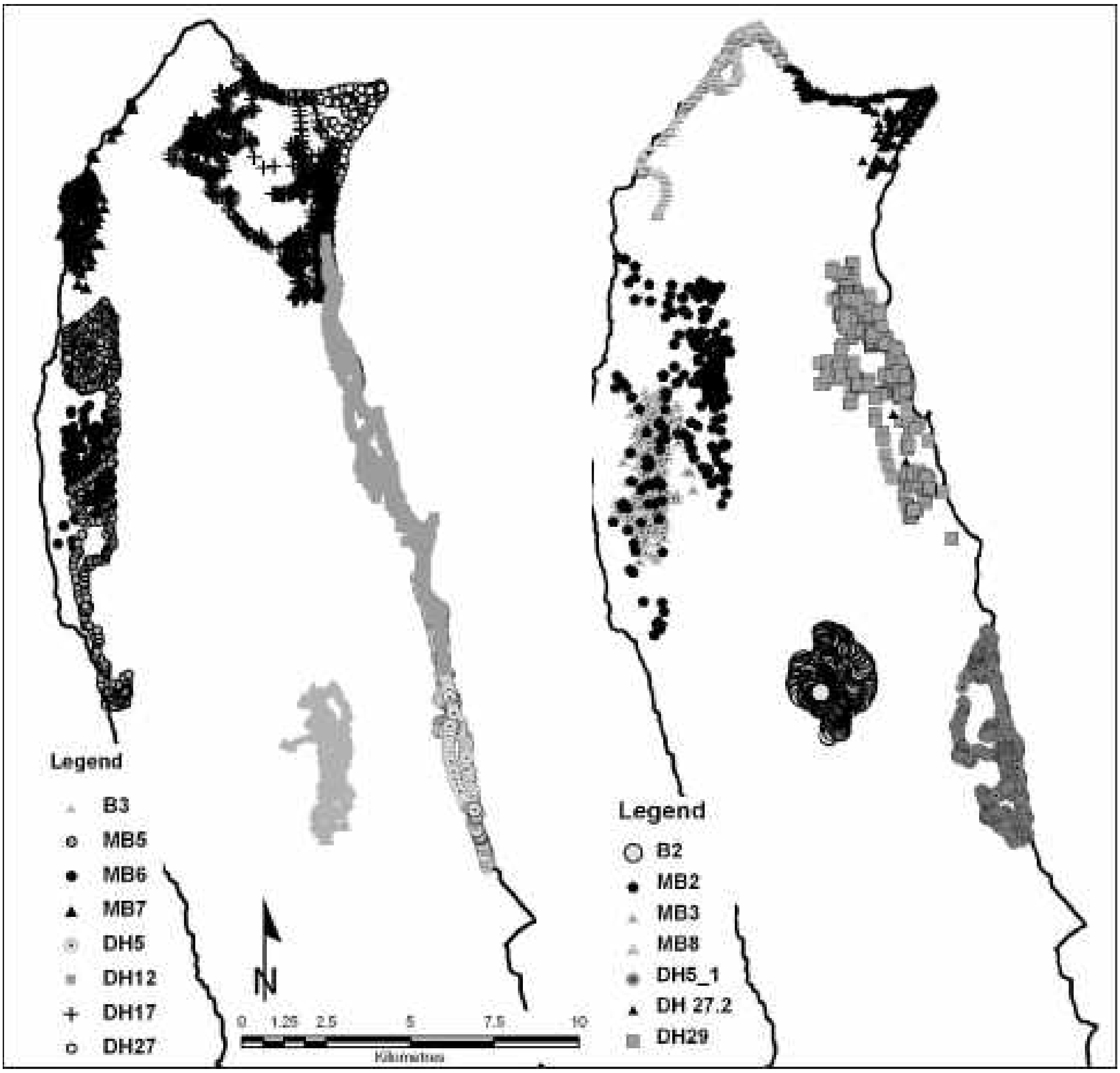

Fig. 4 Compilation of all data (HDOP > 10) derived from GPS data-logger collars fitted to feral cats between 25 March and 5 May 2009. Two maps were necessary due to the high overlap of individual cats.

a transect for individual cats is presented in Table 2. Two cats, MB5 and MB6, had a low likelihood of being detected at all on the 2000 m spacing (7.5% and 18% respectively).

Impact of baiting programme

When baits were spread on 19 April, 15 collared cats were known to be alive. A GPS data-logger on cat (B1) indicated that it moved out of the study area and died before baits were applied. Ten collared cats died after consuming aerially delivered *Eradicat* baits and nine of these had Rhodamine B stains throughout their gastro-intestinal tracts. The tenth animal did not show any Rhodamine B stains but displayed typical symptoms of death from 1080 poisoning. It is not possible to determine whether this cat moved out of the zone where baits containing the Rhodamine B HSDV had been applied or rejected it during feeding as the collar had ceased collecting data. Cats B2 and B3 died following consumption of *Eradicat* bait(s), probably as a result of baits that were distributed by hand in their vicinity on 27 April. The mortality signal from both collars activated 24 hours after bait application by hand and both carcases indicated 1080 toxicoses. In total, 80% of the radio-collared feral cats consumed a toxic bait. Cats DH27, DH29 and MB8 were shot to recover their collars after they had not consumed baits by 1 May (i.e. 12 days post aerial baiting). Bait consumption was highest the day following bait application. However, aerially deployed baits remained palatable to some animals at least ten days following application, given that cat DH17 consumed a bait on 29 April.

Surveys of cat activity

Indices of activity declined following bait application at similar magnitude to radio-collar returns. Compared with values before bait application, PAIs were 83% lower after baiting (z = 3.27, P < 0.001), with PAIs (mean ± s.e.) of 0.079 ± 0.019 and 0.013 ± 0.006 recorded for pre- and post-bait surveys respectively. Similarly, there was a significant difference (t = 6.96, P < 0.001) in the TCIs following baiting with >90% reduction recorded. TCIs pooled over transects recorded 2.75 ± 0.34 cats/10 km transect prior to baiting and 0.25 ± 0.09 cats/10 km transect following baiting.

DISCUSSION

The trial indicated that 10/15 cats died after the aerial spread of baits and a further two animals died after eating baits distributed by hand. Furthermore, reduced indices of activity indicated >80% of the feral cat population died following bait consumption. Our results demonstrate that *Eradicat* baits spread by air will be effective as the primary knock down technique for an eradication campaign on Dirk Hartog Island. During this study, prey for cats appeared plentiful; an abundant rodent population likely related to significant rainfall events over the past two years. Additionally, several collared cats were also implicated in predation of loggerhead turtle (*Caretta caretta*) hatchlings (Hilmer *et al.* 2010). Even greater baiting efficacy might have been achieved when prey was less abundant as optimal rates of bait consumption by feral cats are achieved during periods of food stress (Short *et al*. 1997; Algar *et al*. 2007). The actual eradication will be timed for a period of minimal prey availability.

Bait consumption is not only a function of their attractiveness and palatability but also their accessibility (Algar *et al*. 2007). All cats in this study should have had some opportunity to encounter baits given the baiting intensity and pattern flown by the aircraft. Despite being opportunistic predators, cats will only consume a food item if they are hungry (Bradshaw 1992); if a bait is encountered when the animal is not hungry it may not be consumed regardless of the attractiveness of the bait. Therefore baiting intensity and distribution pattern as well as bait longevity are critical components of successful baiting campaigns. Increasing baiting intensity beyond 50 baits km^{-2} along 1.0 km flight path widths will not necessarily improve baiting efficacy (Algar and Burrows 2004). Analysis of daily cat movement patterns on the island and encounter rates for various transect spacings suggest that reducing flight path spacing to 0.5 km may result in increased bait encounter, particularly in the short-term and may put more cats at risk. Cats B2 and B3 were presumed to have consumed baits applied by hand on 27 April given their patterns of behaviour in the period following application of aerial baits (they were readily located in similar positions during daily checks between the 21 and 27 April). The home ranges inhabited by these cats, in particular B3, were centrally located between aerial bait transects and as a result these animals had less opportunities to encounter a bait. These cats would have encountered baits more often if the flight lines were at intervals of 0.5 km rather than 1.0 km.

All three cats that survived the baiting campaign were in excellent body condition and were obviously not food stressed. Two of these animals occupied/patrolled beaches while the remaining cat probably used other food sources as it was not thought to be accessing beaches where turtle hatchlings were available. All three animals frequented one or more ‘bait exclusion zones’ but also spent time where baits were present. The proposed eradication plan will seek exemption from the requirement to establish ‘bait exclusion zones’, as these may provide a bait-free refuge for cats, particularly those with small home ranges such as juveniles and sub-adults.

Our activity data were biased towards heavier animals, because collars could not be fitted to cats <2.1 kg in weight. The fate of juvenile and sub-adult feral cats following application of baits is thus difficult to assess. GPS data-logger collars were fitted to a larger number of male cats than females for the same reason. Smaller, lighter weight GPS data-logger/radio-telemetry collar are likely to be available in the near future. We proposed to fit these collars to juvenile/sub-adult cats before the eradication programme to test whether our existing strategy places smaller cats at risk. If the collars are still unavailable prior to baiting, this group of cats will be fitted with VHF radio-collars and their survival/mortality monitored following baiting. All animals within the population should be targeted in the eradication programme. The modifications proposed to the current baiting regime should maximise the likelihood of the entire cat population encountering baits.

Most cat ranges were coastal or near-coastal and prey appeared more abundant in these areas. To compensate for this apparent uneven distribution of cats we propose to provide additional baits in more complex topography such as that around the coast.

Baiting alone is unlikely to eradicate the feral cats, so an intensive monitoring and trapping campaign will also be conducted to remove survivors. Placement of monitoring transects will strike a balance between limiting vegetation disturbance and erosion and optimising cat encounters during proposed survey periods of two weeks duration each month. Cat movement data suggest that monitoring transects 1.5 to 2.0 km apart would be sufficient to enable detection of adult animals within each survey period. Data obtained from radio-collared juveniles/sub-adult cats before the eradication programme will verify the suitability of this transect placement across the population.

The size of the island, in particular its length, poses logistical constraints to simultaneous eradication across the entire island. Because it is not practical or feasible to monitor cat activity over such a large area, we propose to conduct the eradication campaign in two stages over three years from January 2011 – January 2014. The first year would be dedicated to installing infrastructure including monitoring transects and an east-west cat-proof barrier fence. We then propose to conduct the baiting and follow-up monitoring/trapping programme against feral cats from the southern fenced section in 2012. This will be followed by the same exercise in the northern fenced section in 2013.

The estimated cost of the feral cat eradication and independent confirmation of success is AUD 2,000,000 excluding salaries over the three year period. Globally, the Dirk Hartog project could become the largest island from which feral cats have been eradicated.

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baiting programme a DEC "1080 Risk Assessment" was undertaken, and approval from the Australian Pesticides and Veterinary Medicine Authority granted (Permit Number 10634v4).

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A review of small Indian mongoose management and eradications on islands

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Abstract The small Indian mongoose (*Herpestes auropunctatus*) is one of the world's 100 worst invasive species. It is a generalist feeder blamed for many declines and extirpations of vertebrates on islands. Native to Asia, it has been introduced to at least 64 islands (Pacific and Indian Oceans, Caribbean and Adriatic Seas) and the mainland (Europe, South America, Australia and North America). Most introductions were in the late 19th and early 20th centuries to control rats in sugar cane fields, but also to control snakes. Although recent mongoose introductions are few, the risk of intentional or accidental spread remains high, and many island taxa are susceptible to their effects. The mongoose has been eradicated from at least six islands (≤115 ha: Buck, Fajou, Leduck, Praslin, Codrington and Green) by trapping and secondary poisoning, but eradication has proven challenging. Two earlier island eradication campaigns against mongoose failed on Buck (182 ha) and Piñeros (390 ha) and campaigns are currently underway on the large islands of Amami-Oshima and northern Okinawa. Attempts to control the mongoose were numerous in the past, and several programmes are underway using trapping and/or poisoning. New techniques are being developed and show promise for eradication. The mongoose can be eradicated with current approaches on small islands with the aim of benefiting endemic species or preventing further introductions. More efficient methods and strategies are needed for successful eradication on larger islands and may facilitate containment of mongoose on the European and South American mainlands.

Keywords: *Herpestes auropunctatus*, invasive, predator, control

INTRODUCTION

Native to the Middle East and much of southern Asia, the small Indian mongoose (*Herpestes auropunctatus,* hereafter mongoose) (Hodgson 1836; Veron *et al.* 2007; Patou *et al*. 2009) has been introduced successfully to islands in the Pacific and Indian Oceans, the Caribbean and Adriatic Seas, and to continental South America and Europe, but was unsuccessfully introduced to North America and Australia (Nellis and Everard 1983; Nellis 1989; Nellis *et al.* 1978; Barun *et al*. 2008). Most introductions were in the late 19th and early 20th centuries to control rats in sugar cane fields, but with questionable success as rat population estimates remained high (Hinton and Dunn 1967). The mongoose was also introduced to control native poisonous snakes including a pit viper, the habu (*Trimeresurus flavoviridis*), on several islands in Japan, the fer-de-lance (*Bothrops lanceolatus*) on Martinique and St. Lucia, *B. caribaeus* in the West Indies, and the horned viper (*Vipera ammodytes*) on Adriatic islands.

The mongoose is a generalist predator that has been identified as one of the world's 100 worst invasive species (IUCN 2000) because of its role in the decline and extirpation of native mammals, birds, reptiles, and amphibians (Baldwin *et al.* 1952; Pimentel 1955a; Seaman and Randall 1962; Nellis and Everard 1983; Nellis and Small 1983; Coblentz and Coblentz 1985; Nellis 1989; Case and Bolger 1991; Henderson 1992; Yamada 2002; Powell and Henderson 2005; Henderson and Berg 2006; Hays and Conant 2007, Barun *et al.* 2010). In their review of the effects of mongoose on native species, Hays and Conant (2007) found that greatest impacts were on native fauna with no past experience with predatory mammals. In addition, the mongoose carries human and animal diseases, including rabies and human *Leptospira* bacterium (Pimentel 1955a; Nellis and Everard 1983).

Eradication of introduced mammals is a powerful conservation tool (Genovesi 2007), but mongoose eradication has been attempted on few occasions and with limited success. A known total of eight eradication campaigns and many control campaigns have been conducted to remove or reduce island mongoose populations. However, even with their limited scope, these attempts probably prevented further declines or even extirpations of native species, although definitive data are lacking. Very few teams have the technical expertise to remove mongoose successfully, even from small islands. Such lack of expertise is reflected by past failures and little progress beyond local control programmes. In addition, most control and eradication efforts are published in the grey literature, if at all, so information is often hard to find for conservation practitioners contemplating mongoose eradication.

We reviewed data from the published and grey literature on eradication and control campaigns, focusing on assessing successes, failures, and challenges. We compiled a list of all islands with known mongoose populations and communicated with researchers and managers who work either directly with the mongoose or with species it affects. Our aim was to facilitate mongoose eradication efforts and direct researchers to areas of applied research that would aid this goal.

BIOLOGY OF THE MONGOOSE

The mongoose is entirely diurnal (AB pers. obs.) and can swim and climb trees (Nellis and Everard 1983), but rarely does so. Mongooses avoid water when possible; they reduce their activity during rainy periods and will not voluntarily enter water deeper than about 5 cm (Nellis and Everard 1983). Such characteristics may account for the failure of mongoose to invade islands only 120 m from occupied sites (Nellis and Everard 1983). However, in Fiji, mongooses get fish out of nets in the water (Craig Morley pers. obs.). This may be a behavioural adaptation specific to that site.

Mongoose home ranges average 2.2 - 3.1 ha for females and 3.6 - 4.2 ha for males; home ranges often overlap and can be as small as 0.75 ha (Nellis and Everard 1983). Areas in the Caribbean may harbour 1-10+ mongoose/ha (Nellis 1989), but populations generally average 2.5 individuals/ha (Pimentel 1955a). On O'ahu, Hawai'i, mean home ranges were 1.4 ha for females and five males shared a region of about 20 ha (Hays and Conant 2003).

Pages 17-25 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. *Island invasives: eradication and management.* IUCN, Gland, Switzerland.

Females are pregnant from February through August in Fiji (Gorman 1976b), the US Virgin Islands (Nellis and Everard 1983), and Hawai'i (Pearson and Baldwin 1953), but the mongoose on Grenada has a 10-month breeding season (Nellis and Everard 1983). Gestation takes 49 days, with litter size of 2.2 on average (range = 1 – 5) (Nellis and Everard 1983). The number of litters produced annually has not yet been determined. Pups begin accompanying their mother on hunting trips at six weeks of age (about 200 g body mass). The youngest wild-caught pregnant female was four months old (Nellis and Everard 1983).

STATUS OF MONGOOSE POPULATIONS

Previous eradication attempts

Globally, at least 64 islands harbour introduced mongooses (Table 1), which are also on the northeastern coastal fringe of South America (Guyana and Surinam; Nellis 1989) and in Adriatic Europe (Croatia, Bosnia and Herzegovina, Montenegro; Barun *et al.* 2008).

Mongoose have been eradicated from six islands and were prevented from establishing on mainland North America when the first few immigrants were caught on Dodge Island, Florida. On Praslin Island, one mongoose was caught in a baited box trap (Dickinson et al. 2001, Quentin Bloxam pers. comm.). The Virgin Islands Division of Fish and Wildlife eradicated a breeding population of mongooses in the 1970s from Leduck Island using 19 x 19 x 48 cm Tomahawk box traps with meat bait (Nellis 1982) and another population from Buck Island in the 1980s also with box traps. This latter success followed an earlier failed attempt (see below). Buck Is has since remained free of the mongoose (McNair 2003; David Nellis pers. comm.).

A campaign on the French West Indian possession of Fajou Island used box-trapping for mongooses and possibly secondary poisoning from a simultaneous rat (*Rattus rattus*) and house mouse (*Mus musculus*) eradication effort using 50 ppm bromadiolone paraffin baits (Lorvelec *et al*. 2004). All trapped mongooses were dissected and none showed toxic bait in the stomach or haemorrhagic syndrome. During a one-month campaign in 2001, 18 people worked full-time to eradicate these three species.

The Antiguan Racer Conservation Project eradicated very small mongoose populations from two islands off Antigua in the West Indies. On Codrington Island, mongoose were eradicated using secondary poisoning from ingesting rats (*Rattus rattus*) poisoned with brodifacoum. The bodies of two poisoned mongooses were found (likely the total number that had been present on this very small island). There is also anecdotal evidence that mongooses were present on Green Island at least one year prior to the rat eradication but were absent afterwards. However, no mongoose carcasses were found during the rat eradication campaign (Jennifer Daltry pers. comm.).

In 1976, the US Fish and Wildlife Service received reports of a mongoose sighting at the Port of Miami on Dodge Island, Florida. Trapping conducted in the area yielded one young female. Interviews with people in the area revealed that two other mongooses had been killed by vehicles a month earlier (Nellis *et al*. 1978).

Failed mongoose eradications include Isla Piñeros, Puerto Rico, and an early attempt on Buck Island. The latter eradication campaign was initiated by the US National Park Service in 1962 (Everard 1975; cited by Everard and Everard 1992). After 10 years of trapping and poisoning, mongooses remained, and eradication efforts were eventually stopped because the ranger conducting the programme was transferred (Nellis *et al.* 1978, Nellis pers. comm.).

On Isla Piñeros fish baits with thallium sulfate may have killed all adult mongooses, which ceased to appear in traps seven days after poisoning began. However, four months later several juvenile mongooses were trapped, indicating that either they had been present in dens, had been too small to spring the traps, and/or bait density had been insufficient to put these juvenile mongooses at risk possibly owing to a reduced home range (Pimentel 1955b).

Current eradication campaigns

We know of only two current island efforts to eradicate the mongoose. Both attempts are in Japan where the mongoose is present on Okinawa and Amami-Oshima in the Ryukyu Islands, and on the main island of Kyushu. The Kyushu population is regarded by some as a recent discovery, but according to locals, mongoose have been there for at least 30 years.

On Amami-Oshima, the Japanese Ministry of the Environment began intensive mongoose control in 2000. Earlier control by local governments of Naze city (1993-2003, 128 km^2), Sumiyo Village (1998-2002, 118 km^2), and Yamato Village (1995-2003, 90 km^2) captured 8,229 mongooses from 1993 until 1999. In an extensive alien eradication programme initiated by the Ministry of the Environment, mongooses were livetrapped by local residents, mainly on a bounty system from 2000 until 2004. Between 60,000 to 317,000 trap-nights and 40 to 131 trappers captured 16,636 mongooses over the five years. The trappers were paid about US$ 20 per mongoose the first year, about US$ 36 the second and third years, and about US$ 45 the last year to try to increase incentives at low abundance. In 2003, three full-time trappers were employed to capture mongooses in low-density areas and began using kill traps. In 2009, 44-48 people were working full-time as Amami Mongoose Busters. Over a five-year period from 2005 until 2009, the Amami Mongoose Busters captured over 7,500 mongooses. From 2000 until 2004 about US$ 1,140,000 (122,000,000 JPY) was spent on the Amami-Alien control programme and from 2005 to 2009 about US$ 7,224,000 (695,000,000 JPY) on the Amami-Mongoose eradication programme (Abe *et al.* 1991; Ishii 2003; Yamada 2002; Yamada and Sugimura 2004; Shintaro Abe pers. comm.). A continuing eradication effort is planned until 2014.

On Okinawa, the Okinawa prefecture and the Japanese Ministry of the Environment initiated an alien control programme (2000-2004) in the Yambaru area of the northern part of the island, and in 2005 this became an eradication campaign. By 2009, 30 people were employed as full-time Yambaru Mongoose Busters. About four km of mongoose-proof fence was constructed in 2005 and 2006 by Okinawa prefecture to separate the trapped area (about 30,000 ha) from the uncontrolled area. From 2000 until 2004, 1831 mongooses were captured with 555,000 trap-nights, and from 2005 until 2009 the Yambaru Mongoose Busters captured over 2680 mongooses with 2,431,000 trap-nights. The total cost for the eradication programme from 2005 until 2009 in the Yambaru area by Okinawa prefecture was about US$ 5,058,000 (486,000,000 JPY including fence construction) and for the mongoose eradication programme by the Ministry of the Environment was about US$ 2,352,000 (226,000,000 JPY) (Yamada and Sugimura 2004, Shintaro Abe pers. comm.).

Past and present "control"/management

Adriatic

In Europe, the mongoose is present on the Croatian islands of Mljet, Korčula, Hvar, Čiovo, Škrda, and Kobrava, as well as the Pelješac Peninsula. The species has recently spread along the coast in Croatia, Bosnia and Herzegovina, and Montenegro at least as far as the Albanian border (Barun *et al.* 2008, Ćirović *et al.* 2011), but the full extent of the range is unknown. The coastal spread of mongoose may have resulted from several separate introductions. Two private mongoose control campaigns are being conducted by local hunters on Hvar and on Čiovo. On Hvar, under the guise of predator control, hunters are required annually either to pay a fee (equivalent to C. $US100) or to submit three mongoose tails or one tail of a native stone marten (*Martes foina*). Most mongooses are trapped there in locally made cages or leg-hold traps. On Čiovo, the only Adriatic island with the mongoose and not the stone marten, the regional hunting organization distributes "rat" poison for mongoose control during the annual autumn meeting (this procedure is illegal in Croatia, so we could not determine which poison).

Caribbean

In the Caribbean, the mongoose is present on 33 islands, many of which have no control (Table 1). Of the occupied islands in the British Virgin Islands, only Jost Van Dyke (JVD) has ongoing mongoose control. The mongoose was introduced to JVD in the 1970s to get rid of the rear-fanged colubrid snake (*Borikenophis portoricensis*). In 2006, the JVD Preservation Society with the help of several volunteers started live-trapping mongooses (Susan Zaluski pers. comm.).

In Puerto Rico, the US Forest Service and USDA APHIS Wildlife Services livetrapped in El Yunque National Forest to protect the critically endangered Puerto Rican parrot (*Amazona vittata*). The US Forest Service annually spends about $10,000 a year with two personnel who trap periodically, so the cost for mongoose control alone is difficult to estimate. A scheduled control of rabies virus vectors was planned for 2010, and targets included the mongoose (Everard and Everard 1992; Pimentel 1955b; Felipe Cano pers. comm.).

In Jamaica, the Jamaican Iguana Recovery Group collaborated in 1997 with Fort Worth Zoo, Milwaukee County Zoo, Zoological Society of San Diego and the University of the West Indies, Mona, to initiate a mongoose control operation in the central Hellshire Hills to protect the critically endangered Jamaican iguana (*Cyclura collei*). Live traps are operational every day and >1000 mongooses have been trapped to date. The approximate cost is US$ 400/month for the salary for one person (Byron Wilson pers. comm.). Two islands near Jamaica, Goat Major and Goat Minor, have been proposed for simultaneous eradication of mongooses and cats, in addition to goats.

On the US Virgin Island of St. Croix, USFWS conducts small-scale mongoose control near sea turtle nesting sites during the turtle breeding season at Sandy Point National Wildlife Refuge (Claudia Lombard pers. comm.). Tomahawk traps are used along 200 to 500-m lines along the beach vegetation. A similar mongoose trapping programme by Virgin Islands National Park staff has been ongoing for five years on St. John. Mongooses are livetrapped on beaches at Hawksnest, Dennis, Jumbi, Trunk, Cinnamon, Maho, Francis, Leinster, Coccoloba, Western Reef Bay, Genti, Little Lameshur, Great Lameshur, and Salt Pond Bay; salt ponds; the National Park Service visitor center, and along some roadways on the north shore (Carrie Stengel pers. comm.).

On St Lucia, the Durrell Wildlife Conservation Trust and St. Lucia Forestry Department (Ministry of Agriculture, Lands, Forestry and Fisheries) conducted two short removal experiments using live traps with chicken bait at an iguana nesting site (Matt Morton pers. comm.).

In 1902, the Agricultural Society on Trinidad started a bounty system of paying per carcass turned in; 30,895 mongooses were turned in from 1902 to 1908 and 142,324 from 1927 to 1930. We do not know when the bounty system stopped operating (Urich 1931).

In 1977, between July and December, a mongoose control operation performed by the Public Health Agency on Guadeloupe yielded 15,787 mongooses (Botino 1977 in Pascal *et al.* 1996), but the capture technique details are unknown because all mongooses were submitted by local residents.

On Cuba, nation-wide mongoose rabies control was undertaken between 1981 and 1985. In the municipality of Arabos, Matanzas Province, in 1984, the mongoose control was carried out by injecting 1,161,682 eggs with strychnine sulfate. Eggs were placed in bamboo or tin pipes to protect them from other animals. Non-poisoned baits were used in mongoose traps that were spaced about 30 m apart over an unknown area. Five to ten people worked per team for a total of about 500 people during that entire operation (Everard and Everard 1992).

In the mid-1970s, mongoose rabies control was undertaken throughout Grenada using sodium fluoroacetate (1080) in 50g of glutinous boiled cowhide. Sixteen baiters/trappers and staff using two vehicles distributed about 300 baits per baiter every day for about nine months. Average mongoose densities dropped from 7.4 to 2.5, but within six months the population recovered (Everard and Everard 1992).

Pacific

In the Hawaiian islands, many sightings of mongooses and one road kill in the 1970s were reported on Kauai but none have been trapped recently despite an extensive effort over the entire island. Elsewhere, widespread control or eradication is not being attempted, but mongoose control is performed in many small (<100 ha) areas to protect birds in upland native bird sanctuaries, wetlands, and wet forests during the breeding season. Agencies involved include the US Fish and Wildlife Service, Hawaii Nature Conservancy, Hawaii State Department of Land and Natural Resources (Wildlife Division), US National Park Service, USDA Wildlife Services, (Department of Army) along with private landowners. Live-traps (Tomahawk) and registered (SLN-Hawaii) diphacinone (50 ppm) wax bait (in bait stations) are employed. The US Department of Agriculture on the island of Hawaii has recently completed field studies evaluating various lures, attractants, and bait types (Pitt and Sugihara 2009). Staff performing mongoose control work are also responsible for other duties, so it is difficult to estimate the total cost for the State of Hawaii (Robert Sugihara pers. comm.).

The small Indian mongoose occurs on 13 islands in Fiji, where a recent molecular study also identified some populations of the Indian brown mongoose, *Herpestes fuscus* (Morley 2004, 2007; Patou *et al.* 2009). Currently there are no attempts to eradicate either mongoose species from any of the Fijian islands (Craig Morley pers. comm.).

Table 1 World list of islands separated into geographic areas and mainland areas where the small Indian mongoose was introduced; islands marked + are interconnected; GID # is Global Island Database number for each island; if the status column is empty then there are no known control attempts.

Island	GID #	Country	Area (ha)	Humans	Status	Refs (presence)	Refs (control)
Adriatic							
Hvar	6760	Croatia	29,737	Yes	Hunters trapping	53; 2	2
Korčula	7300	Croatia	27,840	Yes		53; 2	
Mljet	13790	Croatia	9800	Yes		53; 2	
Škrda	129520	Croatia	200	No		53	
Kobrava	240130	Croatia	52	No		25	
Čiovo	28550	Croatia	2900	Yes	Hunters poisoning, low pop, bridge to mainland	53; 2	2
Caribbean							
Jost Van Dyke	58740	British Virgin Is	850	Yes	JVD Preservation Soc traps	40	52
Tortola +	19250	British Virgin Is	5570	Yes		40	
Beef Island	88670	British Virgin Is	372	Yes		40	
Praslin	No	St Lucia	1	No	Eradicated	15	15; 47
Trinidad	1110	Trinidad & Tobago	476,800	Yes		59	54
Antigua	7140	Antigua & Barbuda	28,100	Yes		40	
Codrington	84837	Antigua & Barbuda	0.5	No	Eradicated	26	26
Green	28660	Antigua & Barbuda	43	No	Eradicated	26	26
Nevis	14620	St Kitts & Nevis	9300	Yes		40	
St Kitts	9890	St Kitts & Nevis	16,800	Yes		40	
St Martin	14960	France/Netherl'ds[1]	8720	Yes		40	
Barbados	5200	Barbados	43,100	Yes		40	
Piñeros	170660	US, Puerto Rico	390	No	Failed eradication attempt; no control	46	46
Vieques	11440	US, Puerto Rico	13,500	Yes		40	
Buck Island	389000	US	72	No	Eradicated	38	38; 33; 44
St Croix	8350	US	21,466	Yes	Localised control	40	11
St John	20180	US	5080	Yes	Localised control	40	12; 9
Leduck	75128	US	5.7	No	Eradicated	39	39
St Thomas	16970	US	8090	Yes	Low population	40	
Water Island	18293	US	199	Yes		40	
Hispaniola	210	Haiti/Dom.Rep.	7,648,000	Yes		40	
Carriacou	26610	Grenada	3770	Yes		20	
Grenada	6510	Grenada	34,400	Yes	Rabies control	40	17
Puerto Rico	790	USA	910,400	Yes	Rabies control	40	17; 46; 18
St Lucia	4090	St Lucia	63,980	Yes	Localised control	40	32
St Vincent	6160	St Vincent	38,900	Yes		40	
Cuba	150	Cuba	11,086,100	Yes	Rabies control	40; 3; 4	17
Romano	4030	Cuba	77,700	Yes		3; 4	
Sabinal	----	Cuba	33,500	Yes		3; 4	
Jamaica	660	Jamaica	1,118,960	Yes	Localised control	16	7
Goat Major +	---	Jamaica	200	No		20	24
Goat Minor	174550	Jamaica	335	No		20	24
La Desirade	35740	France, DOM	2,064	Yes		40	
Fajou	18	France, DOM	115	No	Eradicated	28	28; 34
Grande-Terre, Guadeloupe +	2330	France, DOM	63,900	Yes		40	5
Basse-Terre, Guadeloupe	2330	France, DOM	87,570	Yes		40	5
Marie Galante	10280	France, DOM	15,800	Yes		40	
Martinique	2710	France, DOM	112,800	Yes		40	
Africa							
Mafia	5130	Tanzania	39,400	Yes		59	
Grand Comoro	2840	Comoros	114,800	Yes		29; 58	
Mauritius	1970	Mauritius	204,000	Yes	Localised control	30	49; 8

Table 1 continued

Island	GID #	Country	Area (ha)	Humans	Status	Refs (presence)	Refs (control)
Pacific							
Beqa	25200	Fiji	3620	Yes		35; 13	
Kioa	37310	Fiji	1860	Yes		35; 13	
Macuata-i-wai	102480	Fiji	306	fishermen		35; 13	
Malake	84630	Fiji	453	Yes		35; 13	
Nananu-i-ra	111410	Fiji	270	Yes		35; 13	
Nananu-i-cake	127260	Fiji	300	1 family		35; 13	
Nasoata	25		74	1 family		13	
Vanua Levu	980	Fiji	553,500	Yes		35; 13	
Viti Levu	680	Fiji	1,038,700	Yes		36; 35; 13	
Yanuca	134480	Fiji	154	Yes		35; 13	
Druadrua	90100	Fiji	390	Yes		35; 13	
Mavuva	49	Fiji		Yes		35; 13	
Rabi (Rambi)	66040	Fiji	6878	Yes		35; 13	
Hawaii	700	USA, Hawaii	1,043,200	Yes	Localised control	6	51; 48
Kauai	2360	USA, Hawaii	162,400	Yes	Seen 1970s, not since	55; 10	48
Maui	1950	USA, Hawaii	188,700	Yes		41; 19	
Molokai	3700	USA, Hawaii	67,600	Yes		41; 19	48
Oahu	2210	USA, Hawaii	157,400	Yes		42; 19	48
Amami-Oshima	3610	Japan	71,200	Yes	Ongoing eradication	1	1; 56; 57; 23
Okinawa	2630	Japan	227,130	Yes	Localised control	27	50
Kyusyu	330	Japan		Yes	Recent find, but present about 30 years	37	
Ambon	3470	Indonesia	77,500	Yes		19	
Upolu	2680	Samoa	111,500	Yes	Recent intro Aleipata area	31	
New Caledonia	490	New Caledonia		Yes	Recently introduced	45	
MAINLAND							
Guyana	----	South America	Unknown	Yes		40; 21; 22	
Suriname	----	South America	Unknown	Yes		40; 21; 22	
Croatia (incl Pelješac Pen.)	----	Europe	Unknown	Yes	Coastal area, no known control	53; 2	
Bosnia and Herzegovina	----	Europe	Unknown	Yes	Coastal area, no known control	2	
Montenegro	----	Europe	Unknown	Yes	Coastal area, no known control	2, 14	
Florida	----	USA		Yes	Eradicated	43	

References to Table 1. [1]Abe *et al.* 1991; [2]Barun *et al.* 2008; [3]Borroto-Paez 2009; [4]Borroto-Paez 2011; [5]Botino 1977 in Pascal *et al*. 1996; [6]Bryan 1938; [7]Byron Wilson pers. comm.; [8]Carl Jones and Vikash Tatayah pers. comm.; [9]Carrie Stengel pers. comm.; [10]Case and Bolger 1991; [11]Claudia Lombard pers. comm.; [12]Coblentz and Coblentz 1985; [13]Craig Morley pers. comm.; [14]Ćirović *et al*. 2010; [15]Dickinson *et al.* 2001; [16]Espeut 1882; [17]Everard and Everard 1992; [18]Felipe Cano pers. comm.; [19]Hays and Conant 2007; [20]Horst *et al*. 2001; [21]Husson 1960; [22]Husson 1978; [23]Ishii 2003; [24]Hanson 2007; [25]Ivan Budinski pers. comm. [26]Jenny Daltry pers. comm.; [27]Kishida 1931; [28]Lorvelec *et al.* 2004; [29]Louette 1987; [30]Macmillan 1914; [31]Mark Bonin and James Atherton pers. comm.; [32]Matt Morton pers. comm.; [33]McNair 2003; [34]Michel Pascal pers. comm..; [35]Morley 2004; [36]Morley *et al.* 2007; [37]Nakama and Komizo 2009; [38]Nellis 1978 *et al.*; [39]Nellis 1982; [40]Nellis and Small 1983; [41]Nellis 1989; [42]Nellis and Everard 1983; [43]Nellis *et al.* 1978; [44]Nellis pers. comm.; [45]Patrick Barriere pers. comm.; [46]Pimentel 1955b; [47]Quentin Bloxam pers. comm.; [48]Robert Sugihara pers. comm.; [49]Roy *et al.* 2002; [50]Shintaro Abe pers. comm. ; [51]Smith *et al.* 2000; [52]Susan Zaluski pers. comm.; [53]Tvrtković and Kryštufek 1990; [54]Urich 1931; [55]USFWS 2005; [56]Yamada 2002; [57]Yamada and Sugimura 2004; [58]Walsh 2007; [59]Williams 1918

Recently, mongooses were seen in the Aleipata area of Upolu Island, Samoa and in New Caledonia. One male mongoose was captured during initial trapping on Upolo by the Samoan National Invasive Task Team (Mark Bonin and James Atherton pers. comm.). On New Caledonia, a mongoose infestation was recently reported in Nouméa, and two individuals were trapped (Patrick Barriere pers comm.).

South America

The mongoose is present in Suriname and Guyana but we are unaware of control efforts. Previous reports of the mongoose in French Guiana (Nellis 1989) are not supported by recent evidence (Michel Pascal pers. comm.; Soubeyran 2008).

Africa

On the main island of Mauritius, the Mauritian Wildlife Foundation started a control programme in the Black River Gorges National Park in 1988 as part of the Pink Pigeon Project of reintroduction and predator control (cats, rats, mongooses). Year-round control is conducted with 10-12 students, staff, and volunteers. Wooden box traps (live drop traps) baited with salted fish are primarily used, but for elusive individuals a mix of live/kill traps and change of bait is employed. Estimated total cost is C. US$ 20,000 per year (Roy *et al*. 2002; Carl Jones and Vikash Tatayah pers. comm.).

The mongoose was introduced to Grand Comore during the colonial period (Louette 1987), but no control programme has been reported (Michel Louette pers. comm.). We have no information on mongoose control efforts on the Tanzanian island of Mafia, but the presence of mongoose was confirmed in a recent report (Walsh 2007).

ERADICATION METHODS

Traps and baits

Trapping and toxic baiting have been employed for mongoose control and eradication (Lorvelec *et al.* 2004; Nellis 1982; Nellis *et al.* 1978; Pimentel 1955b; Yamada and Sugimura 2004). Hunting is not known to be employed or expected to be effective.

Mongooses appear susceptible to live traps, particularly box traps, which have been the primary method used to control and eradicate the mongoose. However, anecdotal evidence suggests some animals may become trap-shy or are naturally wary and cannot be trapped with this method (Tomich 1969; AB pers. obs.). Padded leg-hold traps have been used successfully in Hawaii for adult mongooses, but juveniles often do not exert enough pressure to trigger traps unless the trigger is very sensitive (James Bruch pers. comm.). Live traps have the advantage that non-target captures can often be released unharmed, but ethical regulations require them to be checked frequently. Kill traps have been used on Okinawa and Amami-Oshima with great success. Recent trials of the Doc250 kill traps in Hawaii demonstrate that they may be more effective than box traps (Peters *et al*. 2011). Kill traps have the advantage that they do not require routine checks except to re-bait/scent or remove carcasses. Where housings around kill traps can eliminate (or reduce to acceptable levels) the risk to non-target species, kill traps would be the preferred trap type. For eradication campaigns, multiple trap and bait/scent types should be considered, as wariness or aversion to one combination may not be transferable to others.

Live traps have typically been deployed on grids. For eradications, at least one trap must be in each home range area, which is a minimum area of 0.75 ha (Nellis and Everard 1983). The successful campaign on Buck Island used box traps on a 50 x 50 m grid (National Park Service 1993), and that on Fajou used a 30 x 60 m grid (Lorvelec *et al.* 2004). As for other species, having key trap locations is more important than having traps spaced perfectly on a grid. GPS-marked trap locations can be reviewed later via GIS and any coverage gaps addressed. Eradication is possible in small-scale campaigns by trapping alone, but this requires significant manpower and resources.

To facilitate trapping, attractants such as varying types of food are often used. Nevertheless, using lures such as scent (glandular, etc), visual signs (feathers or fur), and auditory cues (prey distress/alarm call, or conspecific calls) may prove useful for mongoose removal or detection. Pitt and Sugihara (2009) found that perimeter baiting was effective, but artificial lures were not. Behavioural traits including home range marking, breeding behaviour, and continual hunting for prey (Gorman 1976b; Nellis 1989) suggest that including attractants might increase trapping and detection success.

Toxic baiting was advocated over 50 years ago as a means of increasing efficacy (Pimentel 1955b), yet few major advances have been made with this method. Because mongooses appear to have low selectivity and consume most bait types (Creekmore *et al.* 1994), baiting is likely to be highly effective. Key considerations include toxin type, bait type, baiting density, non-target species, and timing.

For a chemical to be lethal it must have a pathway and be in a sufficient dosage. Different species have different tolerances to each chemical, and this trait is leveraged to minimise risks to non-target species while putting target species at risk (e.g., Murphy *et al.* 2011). Several toxins have been used historically for controlling mongooses, including thallium sulfate, sodium monofluoroacetate (1080), and strychnine sulfate (Pimentel 1955b; Everard and Everard 1992). Mongooses are highly susceptible to diphacinone (LD50 0.2mg/kg BW), a first generation anti-coagulant, and commercial diphacinone bait blocks have been used in Hawaii with mixed results (Stone *et al*. 1994). Diphacinone is currently the toxin of choice for targeting mongooses alone.

Baits used for delivering toxins to mongooses include chicken meat, boiled cowhide, eggs, salted fish, and commercial flavoured blocks (Pimentel 1955b; Everard and Everard 1992). The main problem with using toxic baits for carnivores is that baits typically used to deliver the toxin become unpalatable after a few hours. Baits have been developed for carnivores that remain palatable for >2 weeks for two large-scale programmes. In Texas, a rabies vaccination programme uses bait blocks effectively for multiple species, while in Western Australia a meat sausage bait was used to target cats and foxes (Skip Oertli pers. comm. 2009; http://www.dshs.state.tx.us/idcu/disease/rabies/orvp/; Algar and Burrows 2004). These baits may be effective for mongoose programmes.

An important aspect of any eradication attempt using toxic baits is that bait must be available to every individual. The baiting density to achieve this goal varies depending on many environmental factors. Baiting densities for mongoose have already been investigated (Creekmore *et al.* 1994; Linhart *et al.* 1993; Linhart *et al.* 1997; Pimentel 1955b). A density of 24 non-toxic baits/ha has yielded a 96-97% efficacy rate on populations with 5.84 (±1.04 SE) and 5.75 (±1.04 SE) animals/ha (Creekmore *et al.* 1994). Bait consumption trials can be used to determine appropriate baiting densities required for mongooses in specific situations (Wegmann *et al.* 2011).

Maximising efficacy

Various methods with potential use against populations of mongoose may pose risks to non-target species of conservation, cultural, or social importance. In such cases, risk assessments should identify where mitigation methods may be needed or whether some methods should not be employed. Timing is a potential mitigation measure, as some non-target species may periodically be absent from islands. On some islands, native mammalian predators will complicate eradication. For example, Mafia has the Egyptian mongoose (*Herpestes ichneumon*), the Adriatic islands of Korčula, Hvar, and Mljet have the stone marten (*Martes foina*), and many islands have native rodents.

For other problem species of mammals, toxic baiting has been timed to maximise bait uptake by target species while avoiding times when young are being nursed or targets have restricted ranges. Bait uptake can be highest when the usual sources of naturally available food are constrained (Algar and Burrows 2004; Howald *et al*. 2007). Island-specific plans for mongoose should consider their breeding patterns following the increase in day length (Nellis and Everard 1983). Times when female mongoose are nursing young (and may have restricted home ranges) should be avoided. The young in dens may not contact baits but be sufficiently independent to survive, a likely reason for the failed eradication attempt on Isla Piñeros, Puerto Rico (Pimentel 1955b). Mongooses can breed year-round, so two pulses of baiting at an interval of 9 - 10 weeks are expected to be required. The experience on Piñeros Island indicates that a single pulse of baits can kill all adult mongooses, but independent young in dens survive (Pimentel 1955b). Two pulses of baiting have yet to be tried for the mongoose but have been effective on tropical rodents that also breed year-round. Until a single method can demonstrably remove all animals (like poison operations for rodents), eradication plans for mongoose should include other methods to detect and remove survivors, a procedure currently used for cat eradications (Campbell *et al.* 2011).

Aerial baiting may be the most cost-effective, efficient, scalable, and replicable method, because mongooses forage almost exclusively on the ground, where most bait will fall, and they readily take bait. Aerial baiting has successfully delivered baits to eradicate rodents and cats, reducing costs and overcoming issues with access caused by terrain and vegetation (Algar *et al.* 2001; Howald *et al.* 2007). Hand-baiting could be used inexpensively on a small area to mimic an aerial baiting programme and provide proof of concept.

Feral cats and mongooses are found together on many islands. Controlling or eradicating one and not the other may yield little conservation benefit. Targeting both species simultaneously may be an option. Although mongooses are susceptible to diphacinone, cats are approximately 70 times more resistant (LD50 14.7mg/kg BW; Smith *et al.* 2000; Stone *et al.* 1994), and adult cats typically weigh at least 4 times more than adult mongooses. Diphacinone is thus suboptimal for targeting both species simultaneously. Para-aminopropiophenone (PAPP) is proposed as an alternative toxin for cats and other eutherian mammals such as canids and stoats in Australia and New Zealand as they are highly susceptible compared to most non-target species on islands (Fisher and O'Connor 2007; Marks *et al.* 2006; Murphy *et al.* 2007; Murphy *et al.* 2011; Savarie *et al.* 1983). Although no lethal dose (LD) data currently exists for mongooses, it is expected they would be highly susceptible to PAPP. Even if mongoose were four times more resistant than cats, the smaller body weight of mongooses would offset their relative resistance. Research is required to identify the lethal dose for mongooses, palatability, and the probability of emesis. Encapsulated PAPP, as is being developed for feral cats, would mask any flavor of the active ingredient and reduce the likelihood of emesis (Johnston *et al*. 2011).

Most islands with introduced mongooses are inhabited, so methods will need to be acceptable to the local populace while still being effective enough to ensure eradication. Live traps, and possibly kill traps and toxic bait stations, will be the key methods in urban areas where aerial baiting is typically not acceptable. Tamper-proof housings that eliminate access by children, pets, and non-targets must be developed before kill traps and toxic baits can be used in urban areas. Educating communities to the health risks mongooses pose to humans and livestock (Everard and Everard 1992) may facilitate acceptance of a campaign and the required methods by the community.

As for cats, mongoose eradications will require detection methods to confirm success. Methods for detecting cats can be applied to mongooses (see Campbell *et al.* 2011). Historically, box trapping has been the only detection method used in eradication campaigns. Larger and more complex campaigns will require additional methods and management tools to detect remnant individuals and confirm eradication. Tracking tunnels currently used in rodent eradication campaigns should be trialed for efficacy in mongoose detection. On Amami-Oshima dogs and camera traps are being used to detect mongooses (Shintaro Abe pers. comm.), but we were unable to find assessments of their efficacy.

RECOMMENDATIONS

Research funding for mongoose eradication trials is urgently needed. Baiting density, suitable toxins, lethal dosage and bait palatability vary depending on many environmental and behavioural factors. We encourage mongoose trials at smaller scales that can be replicated over larger areas by aerial baiting. Several islands that harbour the mongoose are small and uninhabited, and they can be used to test methods with limited liability.

The best opportunities for eradicating or containing an alien invasive species are often in sites where an invasion is in its early stages, when populations are small and localized and not yet well established. Priority for eradication should also be given to islands that can serve as sources for introduction to other areas and those that harbour endemic fauna.

At present many islands inhabited by mongoose are too large for eradication. Intensive localized control could benefit species that are at risk until eradication methods are developed. If planned carefully, such control could be done during a period when the mongoose is at most risk.

As more mongoose eradications are attempted, it is important that lessons learned from each attempt (whether successful or unsuccessful) and the skills learned be shared to ensure success of future efforts.

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Rodent eradication to protect seabirds in New Caledonia: the importance of baseline biological surveys, feasibility studies and community support

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Abstract Eradications of introduced rodents are important for the protection of seabirds. This paper reports on a two-year programme funded by the Packard Foundation to remove exotic rodents from seabird breeding islands in the lagoon of New Caledonia. Although many such islands are close to inhabited areas and heavily used by local communities, rarely has the biodiversity been studied or pest management undertaken to protect the native biota. This paper emphasises the importance of three key components of the eradication planning process for eradication projects in the tropical Pacific: an initial site assessment, community involvement and a well-prepared feasibility study. The purpose of these projects is the restoration of seabird populations on islands identified as Important Bird Areas. Local project manager was supported by an international partnership between Pacific Invasive Initiative, Birdlife International Pacific Secretariat and the New Zealand Department of Conservation. This support was directed at increasing the capability and capacity of local communities in eradicating invasive species from islands and maintaining pest free status for the benefit of native biota and the communities.

Keywords: Ship rat, *Rattus rattus*, Pacific rat, *Rattus exulans*, brodifacoum, seabirds

INTRODUCTION

Introduced predators, especially rodents, have negative effects on seabirds (Burger and Gochfeld 1994; Thibault 1995; Rauzon 2007; Jones *et al.*. 2008) and eradicating rodents from islands significantly benefits seabirds breeding populations (Lorvelec and Pascal 2005; Howald *et al.* 2007; Pascal *et al.* 2008). Successful eradications require robust planning (Cromarty *et al.* 2002) and social acceptance by local communities (Boudjelas 2009). This paper presents the strategy used from July 2007 until March 2009 to eradicate introduced Pacific rats (*Rattus exulans)* and ship rats (*R. rattus*) on small islands identified as Important Bird Areas (IBAs) in the North Lagoon of New Caledonia (Spaggiari *et al.* 2007).

The conservation goal of these projects was to restore seabird populations, especially those of the Polynesian storm petrel (*Nesofregatta fuliginosa)* and fairy tern (*Sternula nereis exsul*; an endemic subspecies). The only recorded breeding of Polynesian storm petrel in New Caledonia was on these islands in 1998 (Pandolfi and Bretagnolle pers. comm.). The area also hosts the last breeding population (100 pairs) of fairy tern (Baudat-Franceschi *et al.* 2009). Both of these species are preyed on by *Rattus* species (Hansen 2006; Thibault and Bretagnolle 1999; Pierce *et al.* 2007) so the provision of rodent free islands is likely to be of benefit.

We first describe how biological surveys, early engagement with key stakeholders (notably local indigenous Kanak communities), and a feasibility study allowed us to decide if eradication was the appropriate pest management strategy. The feasibility study also helped us to develop eradication methods that fitted the local context. We then show how social acceptance of the project was achieved through ongoing consultation, information sharing with key stakeholders, and the participation of local community members in fieldwork. Finally, we describe the eradication method that was applied in the field. The benefits of consultation and involvement of local communities combined with a sound scientific and technical methodology are also discussed.

METHODS

Study site

The north western coast of New Caledonia has a tropical climate with an average rainfall of 1159 mm (732 – 1613 mm) (ORSTOM 1981). The study area is a 20 km wide lagoon with 16 small islands ranging in size from 0.5 - 17 hectares situated between 1.5 and 10 km off the coast (Fig. 1). The islands are flat and sandy with a mixture of vegetation, ranging from short herbaceous ground cover through to coastal forest. Three of the islands have protruding rocky areas rising to an elevation of <50 m and one is a single sand bank. All of the islands are uninhabited but are regularly visited by local fishers and are popular places to visit for the local community.

Feasibility phase

The feasibility study (Baudat-Franceschi *et al.* 2008) included three components: 1) an assessment of

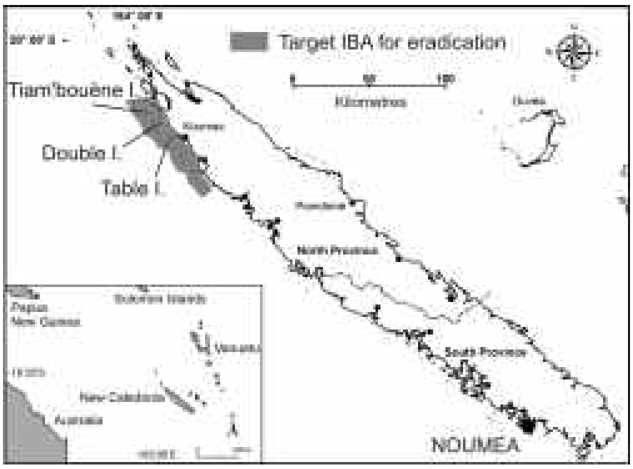

Fig. 1 New Caledonia showing the Important Bird Area with islands from which rodents have now been eradicated.

Pages 26-31 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. *Island invasives: eradication and management.* IUCN, Gland, Switzerland.

technical feasibility of eradicating rodents including non-target species risk assessment; 2) an overview of seabird conservation needs to ensure eradication was the appropriate management choice; and 3) stakeholder issues, because operational feasibility also relies on social acceptance. Biological data were collected from rodent trapping and biodiversity surveys of plants, habitats, hermit crabs, ants, reptiles, and birds. Reptiles and birds were the main non target species. Assessing plants and habitats is necessary to establish the site's ecological characteristics. Ants as a group include some highly invasive species. Hermit crabs are known to consume rodent bait, so assessing their abundance is important for any eradication project (Wegmann *et al.* 2008). Consultation and involvement of stakeholders began during this phase to build social acceptance.

Rodent trapping

Trapping was carried out on eight islands: Ouanne, Pouh, Yan dagouet, Tiam'bouène, Table, Double, Tangadiou, and Magone. The latter two were not considered priority IBA sites but could potentially act as stepping stone islands for rodent invasion between the mainland and Table Island. Because the distance between each island is <2 km, rats could potentially swim from one island to the next (Russell *et al.* 2005). Victor rat traps were deployed for three to five nights consecutively; all traps were sheltered inside corflute tunnels to avoid capturing seabirds. The traps were on grid formed of a transect line every 50 m, along which was a trap every 25 m over the entire area of each island. From night 3 to night 5 (inclusive), Victor mouse traps were deployed between rat traps within the grid, on Table, Tiam'bouène, Ouanne, Double, Yan dagouet, and Pouh Islands. Traps were baited with coconut and peanut butter as late as possible in the afternoon to reduce the likelihood of ants and cockroaches completely removing the bait before nightfall. Trapping was carried out to confirm the presence or absence of rodents on each island and to determine what species were present. The short (3-5 nights) trapping time was in response to the logistical difficulties of surveying on such a large number of islets.

Biodiversity surveys

Land bird surveys on the islands involved point counts (Bibby *et al.* 2000) combined with opportunistic observations. Seabird data came from previous surveys on the islands (Baudat-Franceschi 2006; Baudat-Franceschi *et al.* 2009). A specific focus on breeding phenology was needed to identify in which part of the year baiting operations should take place so as to avoid disturbance of breeding birds. Tropical species of seabirds can have protracted breeding cycles and/or rely on food availability which varies temporally, resulting in significant inter-annual variation of the laying period (Hamer *et al.* 2002). Plant surveys were carried out by Butin (2008) and ant surveys by Le Breton (2008). Reptile diversity was assessed by opportunistic observations during the day and by spotlighting at night. Main terrestrial habitats were mapped using satellite imagery and GPS mapping in the field.

Non-target risk assessment

The information obtained during the biodiversity surveys was used to develop a non-target risk assessment (see Baudat-Franceschi *et al.* 2008). Mitigation measures were incorporated into the eradication design to minimise the risk to non-target species.

Species identified as being potentially at risk from poisoning included non-breeding herons and raptors that occasionally forage on the islands and may scavenge dead rats or prey on hermit crabs: rufous night heron (*Nycticorax caledonicus*), swamp harrier (*Circus approximans*), whistling kite (*Haliastur sphenurus*), and brown goshawk (*Accipiter fasciatus*). Also at risk were shorebirds, raptors and gulls that breed on the islands: beach thick knee (*Esacus magnirostris*), Pacific reef-egret (*Egretta sacra albolineata*), barn owl (*Tyto alba delicatula*) and silver gull (*Chroicocephalus novaehollandiae forsteri*). Four species of vagrant shorebird present in low numbers on the islands and which feed on invertebrates in the littoral zone were: Pacific golden plover (*Pluvialis fulva*), ruddy turnstone (*Arenaria interpres*), sanderling (*Calidris alba*), and wandering tattler (*Heteroscelus incanus*). All passerines recorded on the islands were also at risk because they are insectivorous and frugivorous. Potential risk pathways for all of the above species were through primary and/or secondary poisoning at the individual bird level (e.g., Eason and Spurr 1995; Merton *et al.* 2002; McClelland 2002). However, all were common species and the risk to each at the population level was very low. The exception was the beach thick knee, a shorebird that feeds in the littoral zone. Less than ten breeding pairs have been recorded in New Caledonia, where the species is restricted to the Northern lagoon (Baudat-Franceschi 2006). Because of their small population size, this species was potentially at risk at the population level.

Mitigation measures developed to ensure that risks were minimised included: 1) timing the eradication to avoid the breeding period of most seabirds; 2) the use of bait stations on beaches and other coastal habitats to reduce bait up take by invertebrates that might be eaten by beach thick knee and other shorebirds.

Hermit crab assessment

High numbers of hermit crab have reduced bait availability for target species (Bell 2002; Wegmann *et al.* 2008), so their numbers were assessed for our project. The first assessments were by night walks that followed beaches around the islands. A transect counting system using the rodent trap grid was then used to more accurately assess crab numbers within the site, but especially within vegetation. However, we did not need to systematically cover each site, as it quickly became obvious that were few hermit crabs on the eight targeted islands.

Stakeholder consultation/involvement

Two main stakeholder groups were identified and objectives to achieve support and approval of each group were identified by project manager. The first group comprised the authorities in charge of local environmental legislation. The objectives were to ensure the project was going to comply with the relevant legislation and political will for an approval to be given to carry out the eradication operation. We also wanted to build capacity within administrative departments. This was achieved by providing the authorities with detailed information on the project and its risks, and by holding workshops with those representatives of Northern Province that were responsible for the management of the environment and maritime public domain. These people were also involved in field operations and decision making at each key step of project, such as feasibility/operation/community involvement. The second group was made up of the local community and island users including fishers, tourists, and recreational boat owners. We aimed to ensure that this group was kept well informed about the project to gain their support, and if possible their involvement in field operations. Workshops were held with the Mayor of Koumac and customary authorities. During 23 months of consultation a total of 23 meetings and 14 media events (i.e. radio, television, and newspaper) were undertaken on introduced rodent threats to seabirds, local endangered seabird species, and the broader conservation value of the study area.

Eradication design

Following the feasibility phase, rodent eradication was confirmed as an appropriate management strategy for three islands: Table (11.5 ha), Double (6.5 ha), Tiam'bouène (17 ha). An operational plan was compiled (Baudat-Franceschi 2008) which included the following eradication design: hand broadcast of cereal bait containing 0.02g/kg brodifacoum (trade name Pestoff 20R) over each island in two separate applications except along beaches and the edge of coastal vegetation. At these latter sites, bait stations were used made from corflute boxes that had previously been used to cover traps.

The first application of bait was 13 kg/ha and the second application, which was a minimum of 10 days after the first, was 7 kg/ha. Application of bait was timed to avoid the rodent breeding season and to coincide with the dry season (to avoid bait being washed out by heavy rain). On the three islands, a 20 m wide grid was carefully cut through the vegetation, with bait being broadcast every 20 m (Fig. 2). Bait was spread at each point, by throwing bait in front, behind, to either side and around the feet of the person spreading, a total of five throws. Bait was thrown to a distance of 12 m at a predetermined rate to ensure the island received a complete coverage of bait. The tight grid (20 m x 20 m) and relatively high amount of bait (20 kg/ha) were expected to compensate for thick vegetation and allow for the eradication of mice (*Mus musculus*) in case of previously undetected presence due to short trapping time. The home range of mice is sometimes < 10 m^2 (Faugier *et al.* 2002) and the removal of rats could potentially cause a population explosion of mice ("competitor release effect"; Caut *et al.* 2007). The steep coastal areas on Table Island received double the sowing rate, with baits broadcast from both the top and bottom of the cliffs.

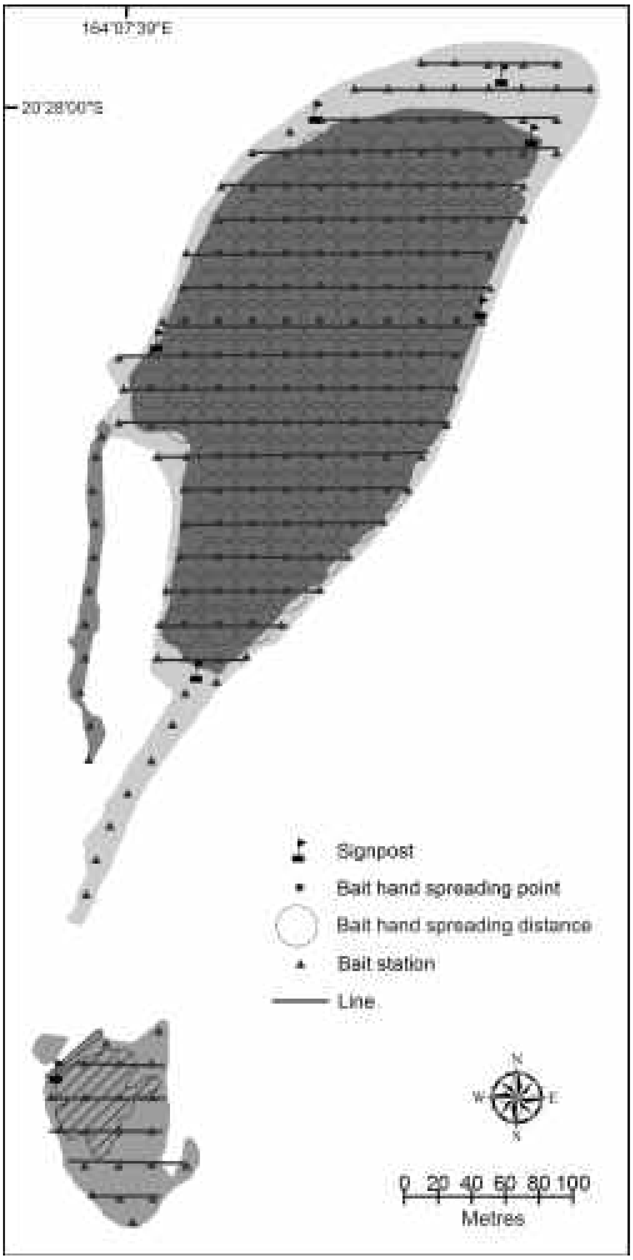

Fig. 2 Double Island showing the pattern of bait stations and hand spreading of bait used on all eradication islands.

Biosecurity

Biosecurity is crucial for long-term eradication success (DOC 2006; Russell and Clout 2007). A biosecurity plan was compiled (Baudat-Franceschi 2009) and approved by local authorities. The plan included an evaluation of reinvasion potential and details of the monitoring systems on each island (e.g., tracking tunnels, permanent bait stations). A reinfestation response procedure and a communication plan for public information (e.g., signposts, media, and flyers) were also included. Additionally, genetic samples from rats of all islands were collected prior to the eradication, so they could be compared with rats found on the islands after the eradication, which will reveal the presence of new invaders or survivors from the eradication (Abdelkrim *et al*. 2005, 2007).

Ecosystem monitoring

Before the eradication began, 20 m x 20 m quadrats were established on each island to monitor plant species diversity and abundance (Butin 2008). Monitoring of species diversity and numbers of seabird breeding pairs, especially those of conservation concern, is being carried out after the eradication. Due to time and funding restrictions, there has been no monitoring of the breeding performance of seabird populations, despite this being a useful indicator of the effectiveness of rodent eradication on seabird populations (Pascal *et al.* 2008).

RESULTS

Rodent trapping

Three priority sites (Ouanne, Pouh, Yan dagouet) and one of the stepping stone islands (Magone) were found to be rodent free. Ship rats were found on Table Island and Pacific rats on Double, Tiam'bouène and Tangadiou Islands.

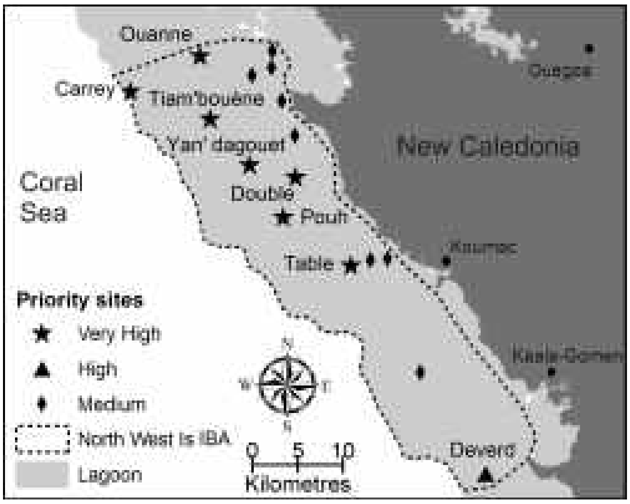

Fig. 3 Islands in the North West Islets Important Bird Area.

Table 1 Results from rodent trapping on eight of the 16 islands in the Important Bird Area.

Island	Area (ha)	Km from coast	Seabird priority	Trapping nights	N trap nights	Captures	Abundance Index*	Species	Trapping period
Tiam'bouène	17	9	high	5	270	51	25	*R. exulans*	March
Ouanne	2.8	7	high	3	63	0		No	March
Double	6.5	4.6	high	4	128	36	33	*R. exulans*	April
Pouh	2.5	6.5	high	3	30	0		No	April
Yan dagouet	4.5	9.1	high	3	57	0		No	April
Table	11.5	5.3	high	4	218	66	39	*R. rattus*	October
Tangadiou	6.5	1.4	low	2	24	5		*R. exulans*	May - July
Magone	1.2	3.2	low	1	17	0		No	July

* Cunningham and Moors (1983): index per 100 trap nights using corrected trap nights number and captures numbers: captures x 100/corrected trap-nights with corrected trap-nights as total trap-nights – trap-nights lost (which is: ½ (captures + sprung, empty traps).

All rat populations had a high abundance index (Fig. 3, Table 1). Populations on three islands were subsequently targeted for eradication. The exception was Tangadiou due to its proximity to the coast (< 2 km) and the ease with which rats might reinvade.

Hermit crabs

Coenobita perlatus was found to be the only species present. Because crab numbers were very low (< 50 individuals per hour of searching), they were considered to represent a low risk of bait interference for this particular project.

Birds

The diversity of land birds breeding on the islands was low (< 10 species per site, Table 2) and there were no ground-dwelling species present. Breeding seabirds included several species of local conservation concern. In addition to Polynesian storm petrel and fairy tern, these included Tahiti petrel (*Pseudobulweria rostrata trouessarti*; an endemic subspecies), wedge-tailed shearwater (*Ardenna pacifica chlororhynchus*), brown booby (*Sula leucogaster plotus*), silver gull (*Chroicocephalus novaehollandiae forsteri*; an endemic sub species), bridled tern (*Onychoprion anaethetus*), crested tern (*Thalesseus bergii cristata*), roseate tern (*Sterna dougalli gracilis*) and black naped tern (*Sterna sumatrana*). Shorebird diversity was low with beach thick knee being the only species breeding on the islands.

Reptiles

Only three species of lizards were recorded, none of which are threatened (Whitaker 2004 and pers. comm.): *Hemidactylus frenatus* (introduced), *Bavayia cyclura sp* (Table Island only), and *Caledoniscincus haplorhinus*.

Table 2 A summation of the species diversity on the three largest islands in the Important Bird Area.

	Tiam'bouène	Double	Table
Native plants	40	43	44
Invasive plants	4	4	6
Total plant spp.	44	47	50
All seabirds	9	9	9
Breeding seabirds	5	4	2
Coastal birds	10	10	10
Land birds	7	8	9
Total bird species	26	27	24
Introduced ants	7	7	9
Invasive ants	1	1	1
Total ant species	8	8	10
N main habitat types	2	6	8

Ants

No sites had any native species of ants but they were all inhabited by the highly invasive tropical fire ant (*Solenopsis geminata*; Table 2; Le Breton 2008). This species is of conservation concern for seabirds, as it can have a negative impact on shearwater chicks (Plentovitch *et al.* 2009).

Plants and habitats

Plant diversity was medium to high, and the flora of each site was of local conservation interest, even though Double and Table islands had significant weed infestations (Butin 2008; Table 2). Native forest predominated on Tiam'bouène Island whereas on Table Island there was a predominance of weeds. Double Island showed an intermediate situation (Fig. 4). The main weed species on these islands were leucaena (*Leucaena leucocephala*) and erect prickly pear (*Opuntia stricta*). Table Island had a dry forest plant of particular conservation concern: an endemic leafflower (*Phyllanthus deplanchei*) (IUCN red listed as Vulnerable). Apart from contributing to planning for the rodent eradications, the work on habitats also helped advocate for future ecological restoration plans for seabird breeding habitats, as part of the IBA overall management plan.

Monitoring of non-target species

Systematic monitoring of non-target species was not developed for this project but searches for the bodies of non-target species after bait spreading failed to reveal any dead non-target species. Beach thick knees, which were thought to be at most risk at from the baits, were observed breeding on Double Island a few days after baiting and continue to do so.

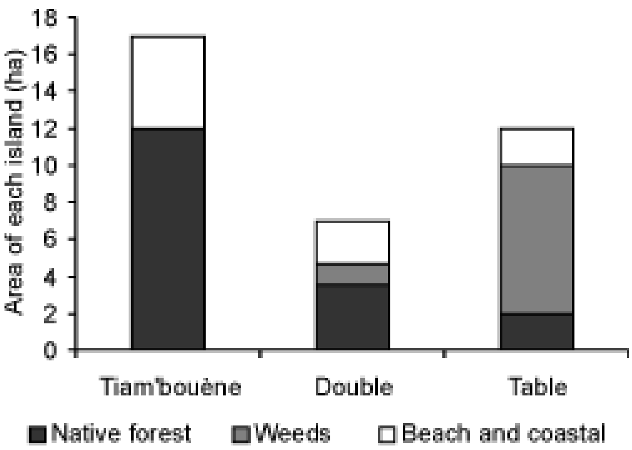

Fig. 4 Comparison of vegetation types on the main islands in this study.

Stakeholder involvement

Local stakeholders were supportive of the proposal to eradicate rodents from the islands and no opposition was encountered. The objectives and methods for the rodent eradication seemed to be widely understood and accepted by the local community. Members of a local kanak non-government organisation "Dayu Biik" were employed in all field operations from initial assessment onwards. Although not kanak tribes of the local Koumac area, Dayu Biik had been trained by an eradication conservation project on Mount Panié. Dayu Biik's members provided a concrete example of skilled people coming from indigenous kanak communities and who are now able to earn money by working within conservation project. They showed people from local tribes one of the long term benefit for their own community of supporting rat eradication (earning money by eradicating rodents). In December 2009, after the eradication, steps were taken by the local Koumac 'big chief' to create an association for island conservation. The Chief would like to involve his community in future seabird and sea turtle conservation by participating in conservation management action, such as guarding endangered seabirds colonies during breeding or by eradicating weeds and rodents.

Eradication outcome

The first application of bait (13 kg/ha) took place between the 1 and 6 September and the second application (7 kg/ha) between the 16 and 19 September 2008. No rat sign was detected on any of the three islands during post-eradication visits undertaken in November 2009 (13 months after baiting).

Implementation of biosecurity measures

Although it is best to implement the biosecurity plan before an eradication operation is carried out, the biosecurity plan for this project is a work in progress, with biosecurity measures being progressively implemented. The incomplete plan is due to timeframe constraints for the project manager and the time needed for the plan to be officially approved by all stakeholders. Although a slow process, development of the plan is promoting long-term co-management of the sites with local communities and other stakeholders. Tracking tunnels and wax tags have been deployed on Double Island and similar deployment is underway on the two other eradication sites. The deployment of permanent bait stations is planned, notably at the Koumac marina on the mainland and on Magone Island.

DISCUSSION

The success of this project can be attributed to collection of baseline information, a well-prepared feasibility study, robust planning, and support from all stakeholders (especially the local community). The partnership between the New Zealand Department of Conservation (NZDOC), Pacific Invasive Initiative and Birdlife International was a decisive element in assisting the local project manager, who had no prior experience in animal pest control. The project manager applied the eradication planning methodology used by NZDOC within the New Caledonian situation.

The feasibility study was one of the first steps of the eradication process and set out to answer three questions: 1) why do the eradication; 2 can it be done; 3) what will it take? Most of the survey information collected fed directly into the feasibility study, and any information gaps were identified at this stage. Carrying out a feasibility study ensured that eradication was an appropriate management objective to help achieve the goal of seabird conservation. In Pacific island countries baseline biological information is often scant or absent. It is important that this information is gathered early in the planning process as it ensures a robust eradication plan is developed. Baseline surveys also added information about the need to manage invasive weeds and pests to fully restore seabirds breeding habitat.

Eradication campaigns can often overlook the need for support from stakeholders, particularly the local community. This project engaged the community from the outset with support from North Province local authorities. This approach has been adopted for other eradication projects in the Pacific (Pierce *et al.* 2007; Wegmann *et al.* 2007) and uses decentralised management by building capacity and capability at the local level (Borrini-Feyerabend *et al.* 2004; Boudjelas 2009). Support for the eradication was built firstly by providing the local community with opportunities to find out about the project and then by involving them in activities like spreading bait and trapping. Although not devoid of difficulties, this involvement helped ensure a successful eradication and also created ownership of the project by the local community. Greater conservation gains can then be made through assistance with ongoing management of the islands. This in turn increases the likelihood of the prescribed biosecurity measures being implemented, through an operational co-management system.

This rodent eradication project is the third one in New Caledonia, following those of Bell (1998) and Caut *et al.* (2009). All have had conservation of seabird populations as the ultimate goal. Our project illustrate that involving the local community was not only a prerequisite for success but also that it greatly improved the capacity to carry out future eradications in New Caledonia.

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Introduced land snails in the Fiji Islands: are there risks involved?

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Abstract Fiji's land snail fauna is highly diverse. There are over 230 species of which about 90% are native and 78% are endemic to the archipelago. There are 18 introduced species and four that are of uncertain origin within the Pacific. Information to allow easy identification of these species is lacking, as is related information about the risks involved with the introduced species in respect to trade, crop production or human and livestock health. To address this latter information gap, existing and new data on Fiji's introduced land snail fauna were collated. This information is urgently required to identify and manage introduced and potentially invasive species and if possible to prevent their spread to non-infected islands. Other Pacific Island countries and territories have suffered substantial endemic land snail biodiversity loss, particularly because of invasive snail species that are not yet present in Fiji. Except for one of these latter species, the giant African snail (*Achatina* (*Lissachatina*) *fulica*), the Fiji government authorities have no baseline reference material that allows them to quickly and accurately identify and understand the biology of even the most common introduced snails. If not addressed this lack of information may have major long-term implications for agriculture, quarantine, trade and human health. The alien species already introduced to Fiji are spreading unacknowledged despite several of them being known disease vectors and agricultural pests elsewhere. This paper provides collated land snail information to government departments such as agriculture, quarantine, forestry and environment, and in turn provides a platform on which to build a stronger understanding of how introduced snail species may be impacting trade, agricultural production and human and livestock health in Fiji.

Keywords: Mollusc, gastropod, slug, Pacific Islands, *Parmarion martensi*, invasive

INTRODUCTION

The land snail fauna of the south Pacific islands of Fiji is unique and highly diverse. Over 230 species are recorded, of which 22 are non-native. About 90% of the fauna is native and 78% are endemic to the archipelago (Barker *et al*. 2005). Information to allow easy identification of species is lacking, as is collated information about the risks non-native species pose to trade, crop production or human and livestock health (Brodie 2009a). Many of the non-native species are known agricultural pests and parasite vectors elsewhere in the world. Collated information is urgently required to detect and adequately manage non-native species, and if possible to prevent the spread of invasive species to non-infected islands.

Pacific Island countries and territories such as Samoa, New Caledonia, French Polynesia and Hawaii (Fig. 1) have lost much of their endemic land snail biodiversity (Bouchet and Abdou 2003; Brescia *et al*. 2008; Cowie and Robinson 2003; Hadfield 1986), in some cases following the introduction of invasive snail species that are not yet established in Fiji. Two such examples are the "rosy wolf snail" (*Euglandina rosea*) and the giant African snail (*Achatina* (*Lissachatina*) *fulica*). Except for the latter species, Fijian government agencies have very little baseline reference material that allows quick and accurate identification of snails. This even applies to the most common introduced terrestrial snails located close to the well established port area of the capital Suva, on the largest island, Viti Levu (Fig. 2). Fijian government agencies also have relatively little collated biological information which could be used to make management decisions or implement monitoring programmes in relation to any of the currently introduced land snail species. If not addressed this lack of information may have major long-term implications for agriculture, quarantine, international trade, and livestock and human health in Fiji.

This current paper is part of a larger plan by the authors to provide direct land snail identification assistance to sectors of the Fiji government such as agriculture, quarantine, forestry and environment, and to improve understanding of how introduced land snail species may impact biodiversity, economic costs and human health in the Fiji Islands. In turn, collation of this information will also allow estimates of the potential impact of these alien intruders on Fiji's established trading partners. In addition, the current paper addresses a broader acknowledged

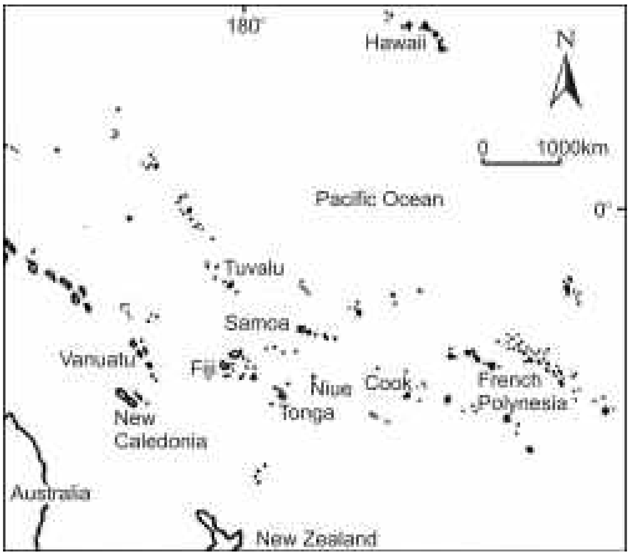

Fig. 1 Fiji's location in the Pacific showing neighbouring islands.

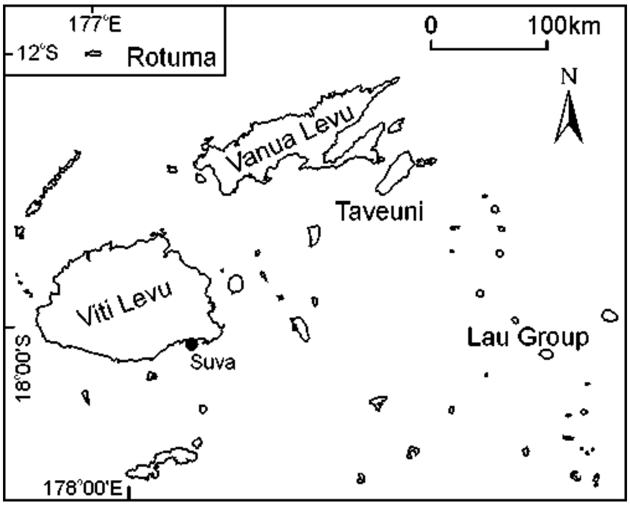

Fig. 2 The Fiji Islands showing the location of the capital city Suva and the islands of Viti Levu, Taveuni and Rotuma. The Lau Group includes all of the small islands in the southeast of the archipelago.

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need to fill major information gaps on the distribution of introduced land snails in the Pacific Islands region (Sherley 2000).

MATERIALS AND METHODS

We compiled a checklist of land snails introduced to Fiji using the results of surveys in many forest areas and villages throughout the archipelago to 2005 (Barker *et al.* 2005; Barker, unpublished data) and in 2008 - 2010 on Viti Levu (Brodie 2009b; Brodie and Copeland in press; Mila *et al.* 2010) and Taveuni (Brodie unpublished data). By combining the above results with our expert knowledge and additional published reports on aspects of distribution, biology, ecology, and “pest” status, we added to our checklist an estimated risk level for each species. Risk level was identified as low, medium or high depending on our estimate of their potential to inflict biodiversity loss, affect agricultural production, and/or impact on human or livestock health in Fiji.

The term ‘land snail’ as a common name is used in preference to distinguishing ‘snails’ and ‘slugs’.

RESULTS

Eighteen species of introduced land snails from nine families are currently known from the Fiji Islands (Table 1). This total excludes the widespread Pacific Achatinellidae *Elasmias apertum*, *Lamellidea pusilla*, *Lamellidea oblonga* and Helicarionidae *Liardetia samoensis* for which precise origins within the Pacific are uncertain.

The feeding types and diets of the introduced species range from herbivores on fresh plant material, detritivores feeding on dead plant material, to carnivorous predators (Table 1). Our data suggest that the introduced *Streptostele musaecola*, *Bradybaena similaris*, and *Deroceras laeve* are restricted to areas of human habitation or disturbance. The remaining species are found in both disturbed and relatively undisturbed habitats and must be considered “invasive”. Of these, nine species are considered here as low risk, three low-medium risk and five medium-high risk (Table 2). One species, *Parmarion martensi* (Fig. 3), stands out as very high risk and very invasive because of its hardy nature, active climbing behaviour, close association with local crops and common presence in virtually all sheltered habitats investigated, including the significant forest conservation areas of Nakauvadra, Nakorotubu and Taveuni. While the presence of *P. martensi* is long-known from Fiji’s lowland to mid-altitude areas, recent surveys by the first author indicate invasion into relatively undisturbed high altitude areas (i.e., Taveuni, > 800 m) that are vitally important for overall ecosystem function and the conservation of endemic biodiversity.

Fig. 3 *Parmarion martensi* on decaying pumpkin in a suburban Suva garden. Photo: G. Brodie.

DISCUSSION

Although many papers have been published about land snails in Fiji over the last 100 years (see review of Barker *et al.* 2005), this is the first to focus on non-native species in the archipelago. The 18 species listed here include several of the expected widespread tropical “tramp” species that are thought to be replacing Pacific Island native/endemic mollusc fauna (Cowie 2004). There is also considerable overlap with the introduced land snail assemblage reported by Cowie (2001) and Cowie and Robinson (2003) in the neighbouring Samoan Islands, but a much lower number of introduced species than the more than 53 species recorded in Hawaii (Cowie 1998; Cowie *et al.* 2008).

Unlike the neighbouring islands of New Caledonia, Vanuatu and Samoa, but like Tonga, Niue and the Cook Islands, Fiji lacks two of the world’s worst invasive land snail species: *Achatina* (*Lissachatina*) *fulica* and *Euglandina rosea*. *Achatina fulica* is a direct economic threat to agricultural production and human and livestock health (Boray 1998; Lowe *et al.* 2004; Raut and Barker 2002), while *E. rosea* poses severe ecological threat by its potential voracious predation on native land snails (Cowie 2001, 2004; Lowe *et al.* 2004).

The risks posed by these two invasive species to Fiji emphasize the need for biosecurity measures to conserve the country’s distinctive and diverse endemic land snail fauna. Lydeard *et al.* (2004) highlighted the global and regional importance of Pacific Island land snail fauna, while Sherley (2000) stressed that “prevention of entry, rather than later control, is the most important means of stopping the spread [and therefore effect] of pest snails”.

In a Fijian context, discussion of the exceptional need for high-level quarantine vigilance is timely, primarily because of the recent nomination of the island of Rotuma (Fig. 2 inset) as a “Port of Entry” for Fijian shipping and trade, but especially agricultural crops. Like many remote islands in the Fijian archipelago, Rotuma has a distinctive land snail fauna (Barker *et al.* 2005; Brodie *et al.* 2010). To the best of our knowledge, no recent survey of introduced land snails has been undertaken either in Rotuma or its intended primarily agricultural trading partner, Tuvalu. In this context the presence or absence of high-risk *Parmarion martensi* in Rotuma and/or Tuvalu is of great interest, not only because of human health concerns and the invasive nature of *P. martensi* in other parts of Fiji, but because the species is also not yet recorded in several countries with which Fiji currently trades, such as Australia, New Zealand and the mainland USA.

Our reporting of *P. martensi* from at least three of the 13 priority forest conservation areas identified on the Fijian islands of Viti Levu and Taveuni (see Olson *et al.* 2009) makes protection of the smaller, more isolated, priority conservation areas like Rotuma an even higher priority.

At least seven of the introduced land snail species found in Fiji act as vectors for parasitic helminthes (Table 2), such as the rat lung worm *Angiostrongylus cantonensis*, which is associated with eosinophilic meningitis in humans (Boray 1998; Hollyer *et al.* 2010). *Angiostrongylus cantonensis* and eosinophilic meningitis are already established in Fiji (Alicata 1962; Sano *et al.* 1987; Paine *et al.* 1994; Uchikawa *et al.* 1984). A recent study of *Parmarion* cf. *martensi* in Hawaii (Hollingsworth *et al.* 2007) identified its role in spreading *A. cantonensis* through an association with poorly washed home-grown crops, such as lettuce. The parasite has a high infection rate and the vigorous climbing behaviour of *P. martensi* makes it much more likely to come into contact with humans (and their food or water sources) than any of the other known vectors. However, the presence of *A. cantonensis* in Fijian *P. martensi* has not yet been confirmed.

Table 1 List of Fiji's introduced land snail species with feeding type and habitat. Feeding ecology, secondary/minor trophic relations indicated in parentheses.

Species	Feeding ecology	Habitat	References
Agriolimacidae			
Deroceras laeve	Herbivore, detrit. (carnivore)	Highland interior, in modified areas, including gardens, and forest margins.	Smith and Stanisic 1998; Barker 1999; Barker and Efford 2004
Ariophantidae			
Parmarion martensi	Herbivore, detritivore	Terrestrial, and arboreal on low vegetation. Lowland to high-elevation forests.	pers. obs., Hollingsworth *et al.* 2007
Quantula striata	Herbivore, detritivore	Leaf litter. Lowland to mid-elevation forests; gardens.	pers. obs., Councilman and Ong 1988.
Bradybaenidae			
Bradybaena similaris	Herbivore, detritivore	Terrestrial, arboreal on low veg. Low to highlands, disturbed areas, incl. gardens.	Pers. obs., Smith and Stanisic 1998; Chang 2002
Pupillidae			
Gastrocopta pediculus	Detritivore	Under stones or logs, in leaf litter. Lowland, in forests and modified areas.	Smith and Stanisic 1998
Gastrocopta servilis	Detritivore	Under stones or logs, in leaf litter. Lowland forests.	Smith and Stanisic 1998
Subulinidae			
Allopeas clavulinum	Detritivore (herbivore)	Leaf litter. Forests and disturbed areas, most prevalent in mid-elevation forests.	Smith and Stanisic 1998
Allopeas gracile	Detritivore (herbivore)	Leaf litter. Lowlands to highlands, in forest and modified habitats.	Smith and Stanisic 1998
Opeas hannense	Detritivore (herbivore)	Leaf litter. Lowlands to mid-elevation forest and disturbed habitat.	Barker *et al.* 2005
Opeas mauritianum	Detritivore	Leaf litter. Lowland to high-elevation forests and distributed area.	Barker *et al.* 2005
Paropeas achatinaceum	Detritivore (herb., carn., predator)	Leaf litter. Lowland to mid-elevation forests and disturbed habitat.	Naggs 1994; Barker and Efford 2004
Subulina octona	Detritivore (herbivore)	Under stones, logs and other debris. Leaf litter. Lowland to mid-elevations forests and disturbed habitat	de Almeida Bessa and de Barros Araujo 1996; Smith and Stanisic 1998; d'Avila and de Almeida Bessa 2005; Juřičková 2006; Hollingsworth *et al.* 2007.
Streptaxidae			
Gulella bicolor	Carnivorous predator	Under stones, logs and other debris. Leaf litter. Lowlands, in forests and modified areas, including gardens.	Annandale and Prashad 1920; Dundee and Baerwald 1984; Naggs 1989; Smith and Stanisic 1998, Solem 1988; Barker and Efford 2004
Streptostele musaecola	Carnivorous predator	Leaf litter, under stones and logs. Lowland disturbed forests.	Smith and Stanisic 1998; Hausdorf and Medina Bermúdez 2003
Veronicellidae			
Laevicaulis alte	Herbivore, detritivore	Under stones, grass, decaying wood, leaf litter & ground crevices. Lowland to high-elevation forests, plantations and moist tall grasslands.	pers. obs., Bishop 1977; Raut and Panigrahi 1990; Smith and Stanisic 1998; Gomes and Thomé 2004
Sarasinula plebeia	Herbivore, detritivore	Under stones, grass, decaying wood, leaf litter and ground crevices. Arboreal on low vegetation. Lowland to mid-elevation forests, plantations, grasslands and gardens.	pers. obs., Bishop 1977; Smith and Stanisic 1998; Rueda *et al.* 2002; Gomes and Thomé 2004
Zonitidae			
Hawaiia minuscula	Prob. carnivorous predator	Leaf litter. Lowland, disturbed areas.	Kano 1996; Smith and Stanisic 1998
Valloniidae			
Ptychopatula orcula	Detritivore	Arboreal, on tree trunks and branches. Lowland forests.	Solem 1964, 1988; Smith and Stanisic 1998

Table 2 Currently known status of introduced land snail species considered to be present in the Fiji Islands archipelago.

Species	Place of origin	Recorded pest/risk type	Where risk recorded	Estimate of risk in Fiji	References
Allopeas clavulinum	Probably tropical East Africa	No known threats	n/a	low	
Allopeas gracile	Probably neotropics	No known threats	n/a	low	
Bradybaena similaris	Asia	Crop pest; vector of human and livestock parasites	Fiji, Australia	medium to high	Alicata 1965; Godan 1983
Deroceras laeve	Holarcic and possibly Andean South America	Crop pest; vector of human and livestock parasites	Australia	low to medium	Mackerras and Sandars 1955; Alicata 1965; Smith and Stanisic 1998
Gastrocopta pediculis	Indonesia. Probably western Pacific-Australian area.	Status unknown could compete with native species	n/a	low	
Gastrocopta servilis	West Indies	Status unknown, could compete with native species	n/a	low	
Gullella bicolor	Indian subcontinent	Predator of native fauna (micro predator on snails)	Australia	medium to high	Smith and Stanisic 1998
Hawaiia minuscula	Canada to northern Mexico	Status unknown, could prey on native fauna	n/a	low	
Laevicaulis alte	Africa	Crop pest; vector of human and livestock parasites	Australia, Hawaii, Samoa	medium to high	Alicata 1965; Malek and Cheng 1974; Liat *et al*. 1965
Opeas hannense	Tropical Central America	Status unknown	n/a	low	
Opeas mauritianum	Unknown, probably India	Status unknown	n/a	low	
Parmarion martensi	South-east Asia	Vector of human and livestock parasites, crop pest	Hawaii, Japan	very high	Hollingsworth *et al*. 2007; Hollyer *et al*. 2010.
Paropeas achatinaceum	South-east Asia, Indonesia	Vector of human and livestock parasites, competes with native species	Hawaii, Pacific Islands	low to medium	Alicata 1965, Cowie 2000.
Ptychopatula orcula	India	Status unknown	n/a	low	
Quantula striata	Southern Malay Peninsula	Status unknown, may compete with native species	n/a	low	
Sarasinula plebeia	Central America	Crop pest; vector of human and livestock parasites	Honduras	medium to high	www.invasive.org; Alicata 1965; Rueda *et al*. 2002
Streptostele musaecola	West Africa	Predator of native fauna (micro predator on snails)	Australia	medium to high	Smith and Stanisic 1998
Subulina octona	Caribbean and tropical America	Crop pest; vector of human and livestock parasites	Brazil, Hawaii	low to medium	de Almeida Bessa and de Barros Araujo 1996; Hollingsworth *et al*. 2007

CONCLUSION

Increased collaborative effort is required to collate and disseminate available land snail information in a user friendly format. Improved access to such information will assist with baseline surveys of isolated priority conservation areas. Although eradication of pest snail species may not be technically possible (Sherley 2000), preventing entry or halting the spread of high-risk pest snails into some countries and islands is more likely to be achieved when local awareness strategies are in place. For the high risk species such as *Parmarion martensi,* these awareness strategies should include provision or reinforcement of the need for preventative public health measures for both local communities and tourist facilities.

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Review of feral cat eradications on islands

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Abstract Feral cats are a substantial threat to native and endemic fauna on islands and are being eradicated with increasing frequency. Worldwide, 87 campaigns have been completed on 83 islands, for a total area of 114,173 ha. Nineteen unsuccessful eradication attempts are known on 15 islands and lessons learnt from those failures are provided. At least five campaigns are currently underway. We review past cat eradication campaigns, and the methods used to eradicate and detect cats in those campaigns. We also review recent advances in eradication and detection methods. We outline proposed eradications and document a trend for increasingly larger islands being considered, but note that although post-eradication conservation impacts are generally positive, there have been some negative ecosystem impacts.

Keywords: *Felis catus*, detection methods, island restoration

INTRODUCTION

While islands make up a small percentage of the earth's total area, they harbour a relatively large percentage of biodiversity, including many threatened species. Islands have also suffered the largest proportion of historic and prehistoric extinctions (Martin and Steadman 1999; Groombridge and Jenkins 2002), many of which are attributable to non-native mammals. On islands, non-native rats (*Rattus* spp.), cats (*Felis catus*), mongoose (*Herpestes auropunctatus*), goats (*Capra hircus*), pigs (*Sus scrofa*) and other introduced mammals have caused localised extirpations, global extinctions and altered ecosystem processes (Coblentz 1978; Ebenhard 1988; Whittaker 1998; Towns *et al.* 2006; Hays and Conant 2007; Jones *et al.* 2008). Feral cats prey on many taxa from invertebrates to large seabirds, and are known to have contributed to over 8% of all bird, mammal and reptile extinctions and to the declines of almost 10% of critically endangered birds, mammals and reptiles (Bonnaud *et al*. 2011; Medina *et al*. 2011). However, invasive species eradication is becoming a well established means of restoring affected islands, with >775 eradications now documented (Keitt *et al.* 2011). Reviews of introduced insular mammal eradications have been published for feral cats, goats, donkeys (*Equus asinus*), mongoose, and commensal rodents (*Rattus* spp., *Mus musculus*) (Nogales *et al.* 2004; Campbell and Donlan 2005; Carrion *et al.* 2007; Howald *et al.* 2007; Barun *et al*. 2011). However, difficulties with collecting unpublished information about eradications and their global scope, mean that reviews typically overlook some eradications. Additionally, the rapid evolution of this field and the increasing rate at which eradications are being conducted mean that reviews are quickly out-of-date. The cat eradication review by Nogales *et al*. (2004) was a landmark paper and has set the stage for future reviews. With insular eradications becoming increasing important to the conservation of biodiversity, we feel that it is timely to update and expand the earlier review to include the numerous additional eradications and technical advances that contributed to their success.

In this paper we review those aspects of cat eradication that will provide useful information for future campaigns. We re-evaluate analyses made by Nogales *et al*. (2004), including island size and eradication methods then add analyses for detection methods. We review new developments in toxicants, baits for aerial spreading of toxicants, and their potential impact on the field of cat eradications. An overview of detection methods that are used to find the last animals and assist in confirming eradication is provided. Of these we highlight preferred techniques. Lastly, we provide an overview of post cat eradication ecosystem responses and recommendations for applied research.

ERADICATION METHODS

Cat eradications have been attempted on islands in all the world's oceans. We found 87 successful campaigns on 83 islands, representing 114,173 ha, that range in size from 5 – 29,000 ha (Appendix 1). We also identified 19 feral cat eradication campaigns that failed on 15 islands (Appendix 2). A further five campaigns are known to be in progress.

Of the 87 successful campaigns, eradication methods are known for 66 (76%). On average, each campaign employed 2.7 eradication methods including leg-hold traps (68%), hunting (59%), primary poisoning (31%), cage traps (29%), and dogs (24%) (Appendix 1).

All successful campaigns for which methods are known on islands >2500 ha (n = 9) utilised primary poisoning with toxic baits, with the exception of Santa Catalina (3890 ha) and San Nicolas (5896 ha). Interestingly, seven failed campaigns on the five largest islands (all >400 ha) for which methods are known did not use toxicants. Toxin use does not guarantee success since five campaigns with toxic baits on four islands <400 ha failed. Of the successful campaigns, 17 campaigns (26% of all) used sodium monofluoroacetate (1080) for primary poisoning. Two campaigns used an unknown toxicant, one campaign used the herbicide paraquat, and another used para-aminopropiophenone (PAPP). Secondary poisoning, leveraged through rodents poisoned with brodifacoum was used in 11 campaigns (17% of all successful), but percent mortality (knockdown) of cats varied. For example, secondary poisoning through eradications of *R. norvegicus* and *R. exulans* was attempted on the New Zealand island of Tuhua, and all cats were removed. However, on Motuihe Island (with *R. norvegicus, Mus musculus*) rabbits were also present, which appeared to be a poor vehicle for transmitting the toxin to cats, and only a 21% population reduction was achieved (Dowding *et al.* 1999; Towns and Broome 2003; P. Keeling pers. comm. 2010). Where rabbits are not present, knockdown rates of ≥80% can be expected for cats when rodents are targeted simultaneously for eradication using brodifacoum. Only three eradications have been completed solely utilising toxin-based methods. In all projects that employ toxins, managers should plan to use other eradication methods to remove remaining animals and capitalise on the population knockdown.

Cage traps have been used with mixed success. Some reports indicate that cage traps were so inefficient at catching feral cats that their use was abandoned in favour of other methods (Domm and Messersmith 1990; Twyford *et al.* 2000; Bester *et al.* 2002). However, cage traps can be useful on inhabited islands where capture and sterilisation of domestic cats is a priority, where domestic cats are non-targets, or where live removal of some animals is a goal. Other traps, such as padded leg-hold live traps are effective at capturing cats and the animals can be dispatched or removed unharmed for sterilisation or live removal (e.g., Hanson *et al.* 2010). Sterilisation of domestic cats on inhabited islands has been used in 8% of all successful campaigns and is being used in two projects that are currently underway (Hilmer *et al.* 2009). Sterilisation of domestic cats is in some cases combined with registration, micro-chipping, legislation or agreements that restrict the importation of cats to sterilised animals or prohibit their importation entirely. Other campaigns, such as on Baltra (Galapagos Islands), utilised agreements to prohibit domestic cats and their importation; pet cats were exported or euthanased.

A relatively new eradication method is fumigation in holes (Springer 2006). The use of aluminium or magnesium phosphide tablets to create phosphine gas that asphyxiates cats in holes may be a valuable method in future campaigns. Cats are highly sensitive to phosphine gas, having a 30 minute lethal gas concentration of 80 ppm, compared to 2400 ppm for rabbits (CDC 1996).

Contrary to claims by proponents of Trap-Neuter-Return (TNR) that it will eventually eliminate cat populations (Longcore *et al.* 2009), feral cats have not been eradicated from any island utilising this technique. There was one unsuccessful campaign where TNR was employed (Appendix 2). Like domestic sterilised cats, neutered feral cats limit the detection methods that are suitable for confirming eradication (e.g., Ratcliffe *et al.* 2009).

We could find cost data for <10% of all successful eradications. To report costs in a single currency, we converted cost data for each year from its native currency to US$ using historical exchange rates for that year (http://fx.sauder.ubc.ca/data.html). If annual cost data were not available, we averaged costs over the years of the campaign. To report costs in a single time period, we adjusted for inflation using historical US annual inflation rates (http://inflationdata.com/). All costs, unless future predicted costs, are expressed in 2009 US dollars (US$). Successful campaigns varied in cost from US$4 – 431 / ha (Table 1).

Feral cat eradication campaigns that we reviewed had a failure rate of 22%. Failures were usually attributed to a lack of institutional support to complete the action, the use of inappropriate methods, and inappropriate timing of those methods. More than half of all successful eradications were on islands <200 ha. Although cats were usually easier to remove from small than large islands, >50% of all known failures were also on islands <200 ha (Appendix 2). Failures on small islands appear to be characterised by a lack of planning and inadequate financial and institutional support. The lack of planning is likely responsible for one of the primary causes of failure: inappropriate timing and methods.

DETECTION METHODS AND CONFIRMING ERADICATION

In addition to the elimination of cats, a second component of eradication campaigns is the use of appropriate methods of detection. Detection methods are crucial to removing the last cats and to determine that the eradication was successful, but these methods have received inadequate attention. Detection methods also help managers determine whether management actions may need modification, such as altering eradication methods, focusing effort in space to remove the last individuals, and gaining insight as to when the last animal may have been removed. In addition, these measures can provide indices of abundance, which are useful for determining the effectiveness of each eradication method employed. Ideally, some detection methods should be independent of eradication methods, so they are not influenced by any aversion induced in the animals. Managers can use detection information, combined with catch-per-unit-effort data from eradication methods to increase confidence that eradication is complete. This approach can also be formalised by conducting detection probability analyses to quantify the likelihood of an animal being detected if present (Ramsey *et al.* 2011).

Detection methods are known from 49 (56% of all) successful cat eradication campaigns (Appendix 1) to search for animals at low densities and to aid in confirming eradication. Commonly used methods were: searching for sign such as footprints, latrines, scat, prey remains (94%), trapping (71%), spotlighting (49%), track pads (43%) and dogs (43%) (Appendix 1). Other methods used were camera traps, baiting, audio and olfactory attractants,

Table 1 The cost (in 2009 US$) of successful insular cat eradication campaigns.

Island	Area (ha)	Cost US$,000s	US$ / ha	Source
San Nicolas	5896	2543*	431*	Island Conservation unpublished data
Wake Atoll (3 isl.)	650	206	317	M. Rauzon pers. comm. 2007
Raoul	2943	832	283	G. Harper pers. comm. (cats and rodents)
Macquarie	12,870	2544	198	S. Robinson pers. comm. 2008
Plata	1420	260	183	Island Conservation unpublished data
Ascension	9700	1300	134	Ratcliffe *et al.* 2009
Mayor (Tuhua)	1277	86	67	Towns and Broome 2003 (cats and rodents)
Baltra	2620	144†	55†	C. Sevilla pers. comm. 2007
Faure	5800	26	4	Algar *et al.* 2010

* Excludes $680,000 in fox mitigation and costs of live removal of cats (A. Little pers. comm. 2010), including these costs the campaign cost $547/ha.
† 47% of total expenditure was spent confirming eradication.

molecular techniques, reproductive status, hair snares and local inhabitants reporting sightings. On average, each campaign employed 3.8 detection methods.

Detection methods most commonly used in cat eradications (Appendix 1) were a combination of searching for sign and an absence of captures in traps. These methods are effective where appropriate substrate allows sign to be easily read and non-target species such as goats, foxes, and seabirds do not confuse or erase sign. In these situations, the probability of detecting sign is increased, trap placement is facilitated, and a paucity of non-target captures allows traps to be available exclusively for cats. Other methods are required where inappropriate substrates exist, or non-target species confound detection. Trappers often create track pads along likely cat travel routes, providing a place in which to later read sign of predictable age and facilitate trap placement. However, track pads are typically informal (a quick smoothing of existing substrate) and often go unreported. Dogs have often been used as a hunting and detection tool. There is great potential in using specialist cat dogs, which have been selectively bred or specifically trained for this purpose (e.g., Wood *et al.* 2002). Camera traps have high rates of detection probability when at appropriate densities and are cost effective when compared to other methods, particularly if substrate is poor for reading sign or when cats are at low densities (Ramsey *et al*. 2011). In a test of several types of camera traps for detecting feral cats, Reconyx Hyperfire No Glow PC900 cameras were competitively priced and had superior battery life, noise and visible light generation, trigger speed and sensitivity, and picture quality (Island Conservation unpublished data). Traps, track pads, camera traps, and hair traps may incorporate visual, auditory or olfactory lures or food baits in an attempt to attract cats.

We recommend that records of the sex and reproductive status of the last animals are kept if these data are available when methods such as trapping are used. Reproductive condition of females is a useful indicator of the presence of males. Foetuses and offspring can be aged (Knospe 2002) to determine whether the last male removed could have sired them. In addition, age of first conception in female cats, which is a minimum of 155 days (Jochle and Jochle 1993), and the presence or absence of uterine placental scars, may be used in a similar way. Further, placental scars may be used to estimate litter size and number of litters in felids (Mowat *et al.* 1996).

Prior to or during an eradication, DNA samples of the population can be collected and stored at little cost. If animals are found after the eradication, samples can then be analysed and microsatellites compared with the original population. This technique may enable determination of whether animals evaded eradication efforts, were introduced, or a combination of these (Abdelkrim *et al.* 2007). Further, DNA analysis can be used to identify individual animals, their sex and determine parent-offspring relationships, which may be important in some situations when dealing with the last animals (Forsyth *et al.* 2005). Blood, tissue samples, faeces and hair with follicles may be used to extract DNA for analysis (Forsyth *et al.* 2005).

The last cat(s) can be difficult to detect, and once detected may be extremely difficult to capture or kill, as was found on Baltra, Raoul, Santa Catalina, Wake and Serrurier Islands (Moro 1997; Phillips *et al.* 2005; A. Cox and B. Wood pers. comm. 2007; Rauzon *et al.* 2008). This highlights the importance of an eradication ethic matched with appropriate techniques and skilled staff to minimise escapes and avoid educating animals (Morrison *et al.* 2007).

Confirming the absence of cats can cost as much if not more than the rest of the eradication campaign (e.g., Baltra, Table 1). An ability to detect cats at low numbers plays a major role in the cost of confirmation and is an area where applied research is needed.

PROPOSED ERADICATIONS

Several insular cat populations are targeted for eradication in the near future. Islands on which cat eradications are in progress include: Robben (507 ha), South Africa; Juan de Nova (440 ha) and Grande Glorieuse (700 ha), France; and Home (95 ha), and West (623 ha), Australia (L. Underhill pers. comm. 2007; Hilmer *et al.* 2009; M. Le Corre pers. comm. 2010). Large islands for which cat eradications have been proposed within the last decade include: Socorro (13,200 ha) and Guadalupe (26,469 ha), Mexico; Floreana (17,253 ha), Ecuador; Auckland (45,975 ha) and Stewart or Rakiura (169,464 ha), New Zealand; and Dirk Hartog (62,000 ha), Western Australia (Beaven 2008; P. McClelland pers. comm. 2009; V. Carrion pers. comm. 2010; L. Luna pers. comm. 2010; Algar *et al*. 2011).

RECENT ADVANCES

Aerial techniques such as bait broadcast and aerial hunting along with the use of GPS and GIS have been of great benefit to rodent and goat eradications over large areas and sites with complex terrain (Campbell and Donlan 2005; Howald *et al.* 2007; Lavoie *et al.* 2007). Second generation anticoagulants have increased the feasibility of rodent eradications (Howald *et al.* 2007). Similarly, aerial baiting techniques against cats provide methods for the rapid knockdown of populations over large areas and complex terrain. The method is enabled by the development of specialist baits for toxin delivery that remain palatable for weeks (Algar *et al.* 2011; Algar and Burrows 2004). The rapid and economical knockdown of ≥90% of a cat population can enable eradications to be conducted in weeks, rather than years (Algar *et al.* 2011; Algar *et al.* 2002). Non-target species may be affected by cat eradication methods or may decrease the efficacy of those methods by consuming bait. Such species increase the complexity of eradications and are a particular challenge. Recent developments in toxins and their applications seek to minimise impacts on non-target species and increase the humaneness of this method. Alternative toxins, such as PAPP, toxicant encapsulation, and exploiting physiological attributes of cats not shared by non-target species, should reduce the risks to other species (Marks *et al.* 2006; Hetherington *et al.* 2007; Murphy *et al.* 2007; Johnston *et al*. 2011). On tropical islands, bait consumption by crabs and decreased palatability from baits being swarmed by ants can pose problems. The use of a residual insecticide, (e.g., permethrin; Coopex, Bayer, Pymble, Australia), which is now integrated into the bait matrix, reduces ant attack while not affecting bait palatability to cats (Algar *et al.* 2007). To reduce non-target bait consumption, a gantry device has been developed that allows cats to access baits but excludes crabs, rats and other non-targets (Algar *et al.* 2004; Algar and Brazell 2008). Baits and leg-hold traps have also been placed on top of buckets filled with sand to reduce crab predation and captures (Ratcliffe *et al.* 2009). Preliminary results from paired food tests indicate that aniseed (*Pimpinella anisum*) may be an effective hermit crab (*Coenobita perlatus*) deterrent (A. Wegmann unpublished data). Further, crabs consuming toxic baits are an additional risk for human populations that consume crab (Pain *et al.* 2008). Future research into compounds for deterring crab consumption of baits could increase the feasibility of conducting cat (and rodent) eradications on tropical islands.

Padded-leg-hold traps such as Victor Oneida # 1.5 soft-catch round-jawed traps are the most commonly used technique in eradicating cats from islands. However, square jawed padded traps provide faster setting, and a greater effective catch area than comparative round jawed traps. Bridger #2 four spring offset custom padded traps are one option and were used effectively on Isla de la Plata. When trap anchors are driven into the ground with wire cable, trappers should use copper ferrules rather than aluminium ferrules to avoid galvanic corrosion, which can result in total decay of ferrules within 21 days, particularly on islands where soils are often high in salts and moisture (Hanson *et al.* 2010).

Leg-hold traps effectively capture feral cats when deployed appropriately (Wood *et al.* 2002), but have the disadvantage of ethical and often legal requirements to check them frequently. Two developments have the potential to fulfil ethical standards while increasing the cost effectiveness of programmes. Telemetry based trap monitoring systems have recently been used on San Nicolas Island to fulfil checking requirements. The trap monitoring system decreased person-hours required to check traps to one-tenth of the effort without the system, and increased animal welfare standards by allowing animals to be removed from traps more promptly (Will *et al.* 2010). Trap monitoring systems can be used for live and kill traps. For small projects, the use of handheld antennae rather than a system of repeaters, as used on San Nicolas, may provide an effective system that will reduce project costs. Trap tabs are small rubber or plastic reservoirs filled with a tranquilising agent and attached to the jaw of a leg-hold trap (Savarie *et al.* 2004). When canines are captured they bite the trap jaw, piercing the reservoir and are sedated, decreasing injury rates (Savarie *et al.* 2004), whereas trapped feral cats do not bite down on trap jaws. Research is underway to develop a trap tab on a throw arm for feral cats that could incorporate a toxicant (e.g., PAPP) or sedative agent (D. Algar pers. comm. 2010). Successful development of this device could provide a humane kill soon after animals were captured, potentially reducing checking requirements.

Specialist cat hunting dogs are a promising detection method, as was indicated by their use on San Nicolas Island (Hanson *et al.* 2010). If required, aversion training can ensure dogs are not a threat to non-target wildlife (Tortora 1982). Furthermore, methods exist to train dogs to avoid toxic baits, and the degradation rate of the compound in baits can be used to determine when it is safe to use dogs in treated areas. Dog tracking by GPS can provide benefits in the field and help managers evaluate terrain coverage of hunters and dogs by GIS. Astro GPS dog tracking units (Garmin, Olathe, Kansas, U.S.) make these activities more economical, but data are frequently lost when there is no line of sight radio signal between the transmitter and handler's GPS. A data saving collar would rectify this problem.

Sentinel cats fitted with radio telemetry or GPS collars incorporating mortality features may be used to monitor the effectiveness of methods (Phillips *et al.* 2005). The capture method for sentinel animals should not bias results. For example, cats captured using bait may be pre-disposed to consuming toxic bait. Blind leg-hold trap trail sets are likely to be the preferred capture method for sentinel animals in most cases. GPS collars can provide additional information on the movements of animals, and potentially alert managers to avoidance strategies being employed by remnant animals.

GIS is possibly one of the most powerful and accessible management tools available for managers of eradication projects. The recent integration of ruggedised handheld field computers with integrated GPS and custom databases facilitated the acquisition, management and interpretation of large amounts of data on San Nicolas Island (Will *et al.* 2010).

Because detecting the last individuals and confirming eradication is so costly for cats, detection probability methods should help managers of future projects to determine stopping rules based on the probability that they would have detected an animal had one been present (Ramsey *et al.* 2011). Furthermore, by combining cost-per-unit-effort with forecasts for maximising detection (and removal) probability from existing data, managers could model each method's cost effectiveness in detecting and removing the last animals and confirming eradication. This would inform decisions about how to deploy the most efficient and cost-effective methods. The incorporation of marked and sterilised cats into the population early in a campaign or before removal methods are applied should improve estimates of probability of detection and removal (Ramsey *et al.* 2011). Data from detection devices can also be used to calculate population estimates (Ramsey *et al.* 2011), and this could be used in near real time throughout a campaign and refined as data becomes available. The development of these management tools will likely only be cost effective for medium-large campaigns until the deployment of these tools becomes more frequent.

The presence of non-target species can influence the selection of methods but trapping techniques have been developed for areas with similar sized non-target carnivores. For example, severe injuries were reduced on endemic foxes on San Nicolas Island when padded leg-hold traps were matched to the size of the non-target species, additional swivels fitted, anchors made as short as possible, and all vegetation that could foul swivels was removed. Walk through sets were identical and a novel scent placed to facilitate recognition and avoidance by endemic foxes; being captured in traps acted as conditioned aversion training. During a 20 day trial, fox captures decreased 95% when comparing the first and last five days, while cat capture rate remained constant (Island Conservation unpublished data). This also demonstrates the risk of poorly set traps, where escape induces aversion to sets. On San Nicolas Island, costs became inflated by restrictions on methods available due to the presence of an island endemic fox (Hanson *et al.* 2010; Table 1). In contrast, although Faure Island is similar in size to San Nicolas, it lacked non-target species that required mitigation or restricted the selection of methods (Algar *et al.* 2011). Cats were eradicated from Faure Island for <1% of the cost of San Nicolas Island (Table 1).

Funding and social issues appear to be the main factors limiting many eradications occurring (Campbell and Donlan 2005; Howald *et al.* 2007), and this is also true for cats. Increasing the efficacy of eradications, particularly confirming eradication, and efficiently implementing multiple species eradications are the primary technical challenges. The use of legislation, spay and neuter, identification by micro-chipping, registration of pets and prohibition or control of importation, will become more common as eradications on inhabited islands involve feral populations of species that are also kept as pets or farm animals (e.g., Ratcliffe *et al.* 2009). Working with communities will be a key component of eradicating cats from inhabited islands. Biosecurity aimed at preventing introductions or reintroductions must also be key components of island management strategies.

POST ERADICATION IMPACTS

Positive responses have been reported for populations of small mammals, reptiles and birds when cats were eradicated (McChesney and Tershy 1998; Donlan *et al.* 2003; Keitt and Tershy 2003; Rodríguez-Moreno *et al.* 2007). Along with increases in extant populations, the creation of introduced predator free habitat can make areas suitable for re-introductions. For example, after cats were eradicated from Faure Island, four species of threatened native mammals that were extirpated by the cats have been successfully re-introduced (Richards 2007). Unassisted recolonisation of species that were extirpated is not uncommon for birds, and often begins soon after cats were eradicated (Schulz *et al.* 2005; Dowding *et al.* 2009; Ortiz-Catedral *et al.* 2009; Ratcliffe *et al.* 2009). Consideration of food web dynamics, and in some cases modelling interactions, may assist in predicting the impacts on conservation targets. For example, Russell *et al.* (2009) modelled rodent-cat assemblages and the impact of eradicating or leaving cats on islands with small long-lived seabirds. Their models suggested that superpredator eradication is crucial for the survival of long-lived insular species. However, cat eradications may also produce unexpected negative ecosystem impacts such as increased predation rates on seabirds (Rayner *et al.* 2007). A report of negative impacts induced by cat eradication on Macquarie Island (Bergstrom *et al.* 2009) was much publicised by the popular press, but several contributing factors were involved and the absence of cats may have been relatively minor among them (Dowding *et al.* 2009).

Before cat eradications are planned, potential positive and negative impacts should be considered in any feasibility analysis. Mitigation actions such as the eradication of other introduced species may also need to be planned. Mixed ecosystem responses to eradication are not restricted to cats (Zavaleta *et al.* 2001; Campbell and Donlan 2005). In addition to considering potential negative impacts on conservation values, managers should also consider the sequence in which invasive species are removed, and plan eradications so that the removal of one species will not complicate or prevent the future removal of another.

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Appendix 1 Successful cat eradication campaigns and the eradication and detection methods employed. In the methods columns a "Y" indicates that the method was used and a "–" indicates that the method was not used or there is no information.

					Eradication Methods													Detection/Confirmation Methods											
Island	Area (ha)	Country	Eradication Period	Year Eradicated	Aerial laid 1080 baits	Ground laid 1080 baits	10/20 Poisoning**	Leg-hold trap	Cage trap	Kill trap	Unknown Trap	Hunting	Dogs	Disease	Domestics neutered	Other	Unknown	Sign	Baiting	Dogs	Trapping	Spotlighting	Molecular	Track pads	Audio attractants	Olfactory attractant	Reproductive status	Other	Unknown
Marion[1,2]	29,000	ZA	1977-91	1991	Y	Y	-	Y	-	-	-	Y	-	Y	-	-	-	Y	Y	-	Y	Y	-	-	-	-	-	-	-
Macquarie[1,3]	12,870	AU	1975-00	2000	-	Y	-	Y	Y	-	-	Y	-	-	-	A	-	Y	-	Y	-	-	-	-	-	-	-	-	-
Tristan da Cunha[4]*†	9837	UK		1974	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Ascension[5]†	9700	UK	2002-04	2004	-	Y	-	Y	Y	-	-	Y	-	-	Y	B	-	Y	Y	-	Y	-	Y	Y	Y	-	-	H	-
San Nicolas[42]†	5896	US	2009-10	2010	-	-	-	Y	-	-	-	Y	Y	-	-	-	-	Y	-	-	Y	-	Y	Y	-	Y	Y	H	-
Faure[6]	5800	AU	2001 (3 w)	2001	Y	Y	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	Y	-	Y	Y	Y	Y	-	-	-
Santa Catalina[7]	3890	MY	2000-04	2004	-	-	-	Y	-	-	-	Y	Y	-	-	-	-	Y	-	-	Y	-	-	-	-	-	-	-	-
Raoul[8]	2943	NZ	2002-05	2005	-	Y	Y	Y	-	-	-	-	Y	-	-	-	-	Y	Y	Y	Y	-	-	-	-	-	-	-	-
Little Barrier[1,9]	2817	NZ	1977-80	1980	-	Y	-	Y	Y	-	-	Y	Y	Y	-	-	-	Y	Y	-	Y	-	-	-	-	Y	-	-	-
Baltra[10]†	2620	EC	2001-04	2004	-	Y	-	Y	Y	-	-	Y	-	-	-	-	-	Y	-	-	-	Y	Y	-	-	-	-	-	-
Motutapu[11]†	2311	NZ	2009	2009	-	-	Y	Y	-	-	-	Y	-	-	-	-	-	Y	-	Y	Y	-	-	-	-	-	-	-	-
Kapiti[1]	1965	NZ		1934	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Monserrat[1, 37]	1886	MY	2000-01	2001	-	-	-	Y	-	-	-	Y	Y	-	-	-	-	Y	-	Y	Y	Y	-	Y	-	-	-	-	-
Monserrat[39]	1886	MY	2002	2002	-	-	-	Y	-	-	-	Y	Y	-	-	-	-	Y	-	Y	Y	Y	-	Y	-	-	-	-	-
Rottnest[12]	1705	AU	2001-02	2002	-	-	-	Y	-	-	-	-	-	-	-	-	-	Y	-	-	Y	-	-	Y	Y	-	Y	-	-
Partida Sur [1,37,38]∆	1533	MY	2000	2000	-	-	-	-	-	-	-	-	-	-	-	C	-	Y	-	Y	Y	Y	-	-	-	-	-	-	-
Rangitoto[11]†	1509	NZ	2009	2009	-	-	Y	Y	-	-	-	Y	-	-	-	-	-	Y	-	Y	Y	-	-	-	-	-	-	-	-
Plata[13]	1420	EC	2009	2009	-	-	Y	Y	-	Y	-	-	-	-	-	-	-	Y	-	Y	Y	Y	Y	Y	Y	Y	Y	-	-
Mayor [1,14]	1277	NZ	2000	2000	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Hermite[1,16]	1020	AU	1999 (8 w)	1999	Y	-	-	Y	-	-	-	-	-	-	-	-	-	Y	-	-	Y	-	-	Y	Y	Y	-	-	-
Alegranza[1]	1020	ES	1997-01	2001	-	-	-	Y	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Trinidade[17]	1000	BR		1989	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Deserta Grande[1]	1000	PT		1984	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Natividad[1,15, 37]†	736	MY	1998-00	2000	-	-	-	Y	-	-	-	Y	Y	-	-	C	-	Y	-	Y	Y	Y	-	Y	-	-	-	-	-
Coronados[1,37]	715	MY	1998-99	1999	-	-	-	Y	-	-	-	-	-	-	-	-	-	Y	-	-	Y	Y	-	Y	-	-	-	-	-
Port Cros[18]†	640	FR	2004-06	2006	-	-	-	-	Y	-	-	-	-	-	Y	-	-	Y	-	-	Y	Y	-	-	-	-	-	H	-
Pitcairn[1]†	500	UK		1997	-	-	Y	Y	Y	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Wake (Wake Atoll)[19]†	500	US	1996-04	2004	-	-	-	Y	Y	-	-	Y	-	-	Y	-	-	Y	-	-	-	-	-	-	-	-	-	-	-
Lobos[1]	430	ES	1992-02	2002	-	-	-	Y	Y	-	-	-	-	-	-	G	-	-	-	-	-	-	-	-	-	-	-	-	Y
Jarvis[1,20]	410	US	1957-90	1990	-	-	-	Y	Y	Y	-	-	-	Y	-	G	-	Y	-	-	-	Y	-	-	-	-	-	-	-
San Francisquito[1,37]	374	MY	2000	2000	-	-	-	Y	-	-	-	Y	Y	-	-	-	-	Y	-	Y	Y	Y	-	Y	-	-	-	-	-
Reevesby[1]	373	AU	1984-90	1990	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Great Dog[1]	370	AU	1991-92	1992	-	-	-	-	-	-	Y	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Santa Barbara[21]	326	US		1950	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Angel[1]	314	US			-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Curieuse[1,22]†	286	SC	2000-1	2001	-	Y	Y	Y	-	-	-	-	-	-	-	-	-	Y	Y	-	Y	-	-	-	-	-	-	-	-
Dassen[23]	273	ZA	mid 1980s-02	2002	-	-	-	-	Y	-	-	Y	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-
San Martin[1,37]	265	MY	1999	1999	-	-	-	Y	-	-	-	Y	Y	-	-	-	-	Y	-	Y	Y	Y	-	Y	-	-	-	-	-
Flat[1]	253	MU		1998	-	-	Y	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Mejía[1,37]	245	MY	1999-01	2001	-	-	-	Y	-	-	-	Y	Y	-	-	-	-	Y	-	Y	Y	Y	-	Y	-	-	-	-	-
Fregate[1]	219	SC	1980-82	1982	-	Y	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Serrurier[25]∆	188	AU	1996	1996	-	Y	-	-	Y	-	-	Y	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-
Motuihe [26]‡	179	NZ	2002-04	2004	-	-	-	Y	-	-	-	Y	-	-	-	D	-	Y	-	-	Y	Y	-	-	-	-	-	-	-
Cuvier[1,27]	170	NZ	1960-64	1964	-	-	-	Y	Y	Y	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Howland [28,29] ∏	166	US		1964	-	-	-	-	-	-	-	Y	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-
Howland[29€]	166	US		1986	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Gabo[1,30]	154	AU	1987-91	1991	-	Y	-	-	Y	-	-	Y	-	-	-	-	-	Y	-	-	Y	Y	-	-	-	-	Y	I	-

Appendix 1 continued

Island	Area (ha)	Country	Eradication Period	Year Eradicated	Eradication Methods: Aerial laid 1080 baits	Ground laid 1080 baits	10/20 Poisoning**	Leg-hold trap	Cage trap	Kill trap	Unknown Trap	Hunting	Dogs	Disease	Domestics neutered	Other	Unknown	Detection/Confirmation Methods: Sign	Baiting	Dogs	Trapping	Spotlighting	Molecular	Track pads	Audio attractants	Olfactory attractant	Reproductive status	Other	Unknown
Desecheo (Puerto Rico)[31]	152	US	1985-87	1987	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Stephens[1]	150	NZ	c. 1910-25	1925	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Denis[1,22]	150	SC		2001	-	Y	Y	Y	-	-	-	-	-	-	-	-	-	Y	Y	-	Y	-	-	-	-	-	-	-	-
Guillou (Kerguelen)[1]	145	FR	1994-95	1995	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Baker[1]	145	US		1930	-	-	-	-	-	-	-	-	-	-	-	E	-	-	-	-	-	-	-	-	-	-	-	-	Y
Moutohora [21]	143	NZ			-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Mou Waho [21]	140	NZ			-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Putauhinu [1]	140	NZ			-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Coronado Sur[39]	126	MY	2003	2003	-	-	-	Y	-	-	-	Y	-	-	-	-	-	Y	-	-	Y	-	-	-	-	-	-	-	-
Tasman[43]	120	AU	2010	2010	-	-	-	Y	Y	-	-	Y	-	-	-	K	-	Y	-	Y	Y	Y	-	-	-	-	-	H	-
Mou Tapu[21]	120	NZ			-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Aziak[21]	118	US		1964	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Mangere [1]	113	NZ		1950	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Long Cay[1,32]	111	UK	1999	1999	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-
Wilkes (Wake Atoll)[19]	110	US	1996-2004	2004	-	-	-	Y	Y	-	-	Y	-	-	Y	-	-	Y	-	-	-	-	-	-	-	-	-	-	-
North West (Capricorn)[1,33]	105	AU	1984-85	1985	-	Y	-	-	Y	-	-	Y	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-
Todos Santos Sur[1,37]	89	MY	1997-98	1998	-	-	-	Y	-	-	-	Y	Y	-	-	-	-	Y	-	Y	Y	Y	-	Y	-	-	-	-	-
Todos Santos Sur[40]	89	MY	1999	1999	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Todos Santos Sur[41]	89	MY	2004	2004	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Estanque[1,37,40]∆	82	MY	1999	1999	-	-	-	Y	-	-	-	Y	Y	-	-	-	-	Y	-	Y	Y	Y	-	Y	-	-	-	-	-
Isabel[1,24]	80	MY	1995-97	1997	-	Y	Y	Y	-	-	-	Y	Y	-	-	-	-	Y	Y	Y	Y	Y	-	Y	-	-	-	-	-
Aride[34]	68	SC		1930s	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Stevensons[21]	65	NZ			-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Yerba Buena[21]	65	US			-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Viwa[35]†Ω	60	FJ	2006	2006	-	-	Y	Y	Y	-	-	-	-	-	Y	F	-	-	-	-	-	-	-	-	-	-	-	J	-
San Jerónimo[1,37]	48	MY	1999	1999	-	-	-	Y	-	-	-	Y	Y	-	-	-	-	Y	-	Y	Y	Y	-	Y	-	-	-	-	-
Anacapa- E[36]	43	US		1970	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Asunción[15,37,40]	41	MY	1994	1994	-	-	-	Y	-	-	-	-	-	-	-	-	-	Y	Y	Y	Y	Y	-	Y	-	Y	-	-	-
Peale (Wake Atoll)[19]	40	US	1996-2004	2004	-	-	-	Y	-	-	-	Y	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-
Matakohe[1]	37	NZ	1991	1991	-	Y	-	Y	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Coronado Norte[15,37]	37	MY	1995-96	1996	-	-	-	Y	-	-	-	-	Y	-	-	-	-	Y	Y	Y	Y	Y	-	Y	-	Y	-	-	-
San Roque[15,37,40]	35	MY	1994	1994	-	-	-	Y	-	-	-	-	-	-	-	-	-	Y	Y	Y	Y	Y	-	Y	-	Y	-	-	-
Todos Santos Norte[40]	34	MY	1999-2000	2000	-	-	-	Y	-	-	-	-	Y	-	-	-	-	Y	Y	Y	Y	Y	-	Y	-	Y	-	-	-
Cousine[1]	30	SC	1985-90	1990	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Farallon S[21]	29	US		1972	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Herekopare [1]	28	NZ		1970	-	-	-	Y	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Ile aux Aigrettes[1]	25	MU		1994	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y
Silver[19]	25	NZ			-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Derbin[19]	15	US		2000	-	-	-	-	-	-	-	-	-	-	-	-	Y	-	-	-	-	-	-	-	-	-	-	-	Y
Hoskyn[19]	5	AU		1979	-	-	-	-	-	-	Y	Y	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	Y

Country abbreviations: AU Australia, BR Brazil, EC Ecuador, ES Spain, FJ Fiji, FR France, MY Mexico, MU Mauritius, NZ New Zealand, PT Portugal, SC Seychelles, UK United Kingdom, US United States of America, ZA South Africa.
Other methods: ** primary/secondary poisoning with brodifacoum; A, fumigation in holes; B, hand capture; C, live removal; D, secondary poisoning from aerial 1080 for rabbit eradication and possibly pindone ground laid baits; E, clubbing w/sticks; F, paraquat herbicide in meat baits; G, ground laid baits with unspecified toxicant; H, camera traps; I, hair traps; J, local inhabitants reporting sightings; K, PAPP baits dispersed aerially and from ground. * domestic cats removed by medical officer due to toxoplasmosis in 1974. Holdgate (1965) reports feral cats over entire island, but none are present today. † inhabited. ‡ cats not eradicated in 1981 as reported by Veitch and Bell 1990. ∆ single cat removed. ∏ cats reintroduced in 1966. € 1966 reintroduction was eradicated in 1986 and is reported in Nogales *et al.* (2004) as 1979 eradication. Ω one male cat hidden by villager, not neutered. No restriction on reintroduction of cats.

Appendix 1 continued

Sources: [1] Nogales *et al.* 2004. [2] Bester *et al*. 2002. [3] Springer 2006. [4] Holdgate 1965; P. Ryan pers. comm. 2007. [5] Ratcliffe *et al*. 2009. [6] Algar *et al*. 2011. [7] B. Wood pers. comm. 2007 and 2009; L. Luna pers. comm. 2010; Aguirre *et al.* 2008. [8] Broome 2009; A. Cox pers. comm. 2007; G. Harper pers. comm. 2010. [9] Veitch 2001. [10] Phillips *et al.* 2005. [11] R. Griffiths pers. comm. 2010. [12] Algar *et al*. unpublished data. [13] Island Conservation unpublished data. [14] Towns and Broome 2003. [15] Donlan *et al*. 2000. [16] Algar *et al.* 2002. [17] R. Valka pers. comm. 2010. [18] Bonnaud *et al.* 2011. [19] Rauzon *et al.* 2008; M. Rauzon pers. comm. 2007. [20] Rauzon 1985. [21] Island Conservation database. [22] Merton *et al.* 2002. [23] L. Underhill and A. Wolfaardt pers. comm. 2007. [24] Rodríguez *et al*. 2006. [25] Moro 1997. [26] P. Keeling pers. comm. 2010. [27] Merton 1961. [28] King 1973. [29] M. Rauzon pers. comm. 2007. [30] Twyford *et al*. 2000. [31] Evans 1989. [32] Mitchell *et al*. 2002. [33] Domm and Messersmith 1990. [34] Parr *et al*. 2000. [35] B. Nagle and C. Morley pers. comm. 2009. [36] K. Faulkner pers. comm.; Knowlton *et al*. 2007. [37] B. Wood pers comm. 2008. [38] M. Hermosillo-Bueno pers. comm. 2010 to L. Luna. [39] Aguirre Muñoz *et al.* 2003. [40] Sánchez Pacheco and Tershy 2000. [41] Aguirre Muñoz *et al.* 2004. [42] Hanson *et al*. 2010; Ramsey *et al.* 2011. [43] Sue Robinson unpublished data.

Appendix 2 Unsuccessful cat eradication campaigns

Island	Area (ha)	Country	Methods	Campaign Year(s)	Reason for failure
Grande Terre (Kerguelen)[1]	650,000	FR	Hunting	1960, 1970-77	Effort ceased once at low numbers (both efforts).
Amsterdam[2]	5500	FR	Unknown	pre 1957	Campaign abandoned when rat and mice numbers increased which was believed to be a response to decreased cat density.
Raoul[3]	2943	NZ	Dogs, hunting	1970s	Caused inefficiency in a concurrent goat eradication campaign and was stopped.
Little Barrier[4]	2817	NZ	Disease, leg-hold traps, cage traps	1968-9	Lack of continuity / insufficient effort.
Plata[5]	1420	EC	Cage traps, trap-neuter-release	2006-07	Inappropriate methods, unable to trap all animals / not all animals at risk.
Jarvis[6]	410	US	Hunting	1964-68, 1973-78	Lack of continuity / insufficient effort / only single technique.
South Molle (Queensland)[7]	380	AU	Ground laid 1080 baits, hunting	1985-86	Staff at the resort hid cats in their rooms. Not all animals were at risk.
Serrurier[8]	188	AU	Ground laid 1080 baits, hunting	1987-90, 1995	Single cat. Failed shooting attempts caused wariness (1st attempt). Abundant food source (breeding seabirds) when baits laid; inappropriate timing (1st and 2nd attempt).
Motuihe (Hauraki Gulf)[9]	179	NZ	Brodifacoum aerial baiting for rodents and rabbits	1997	Complete eradication or knockdown on cat population anticipated by primary/secondary poisoning but only 21% population reduction achieved, possibly as rabbits poor vector for toxin. Funding for follow-up work was unavailable. Inappropriate method / not all animals at risk / lack funding.
Howland[10]	166	US	Hunting, kill traps, cage traps	1977-79	Long grass - hunting ineffective, inappropriate methods didn't put all animals at risk.
Tasman[11, 12] (Tasmania)	120	AU	Ground laid 1080 baits, hunting	1977-80	Seasonal presence of main prey species unknown at the time, contributing to not all cats being vulnerable to baiting. Program halted after 3-4 years effort. Unable to kill animals faster than they reproduced, lack of concentrated effort.
Little Green (Tasmania)[12]	87	AU	Cage traps	1983-84	Inappropriate method. Old cat scat found in December 2007 during a brief visit.
San Roque[13]	79	MY	Hunting	Late 1980s	Campaign abandoned, majority of cats removed. Insufficient institutional support.
Asunción[13]	68	MY	Hunting	Late 1980s	Campaign abandoned, majority of cats removed. Insufficient institutional support.
Wedge (Tasmania)[12]	43	AU	Leg-hold traps, cage traps, hunting, dogs	2003, 2004	Attempted on a limited budget. At the time, eradication not a priority action for the managing bodies, insufficient institutional support for each campaign. Prints and scat present 2008.

Sources: 1 Lorvelec and Pascal 2005; Chapuis et al. 1994. 2 Reppe 1957 cited in Holdgate and Wace 1961. 3 Parkes 1990. 4 Veitch 2001. 5 G. Banda pers. comm. 2007. 6 Rauzon 1985. 7 K. MacDonald pers. comm. 2007. 8 Moro 1997. 9 Veitch 2002; Dowding et al. 1999; P. Keeling pers. comm. 2010. 10 M. Rauzon pers. comm. 2007. 11 Brothers 1982. 12 S. Robinson unpublished data. 13 Donlan et al. 2000; B. Tershy pers. comm. 2010.
Country abbreviations: AU Australia, EC Ecuador, FR France, MY Mexico, NZ New Zealand, US United States of America.

Preparations for the eradication of mice from Gough Island: results of bait acceptance trials above ground and around cave systems

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Abstract Gough Island, Tristan da Cunha, is a United Kingdom Overseas Territory, supports globally important seabird colonies, has many endemic plant, invertebrate and bird taxa, and is recognised as a World Heritage Site. A key threat to the biodiversity of Gough Island is predation by the introduced house mouse (*Mus musculus*), as a result of which two bird species are listed as Critically Endangered. Eradicating mice from Gough Island is thus an urgent conservation priority. However, the higher failure rate of mouse versus rat eradications, and smaller size of islands that have been successfully cleared of mice, means that trials on bait acceptance are required to convince funding agencies that an attempted eradication of mice from Gough is likely to succeed. In this study, trials of bait acceptance were undertaken above ground and around cave systems that are potential refuges for mice during an aerial application of bait. Four trials were undertaken during winter, with rhodamine-dyed, non-toxic bait spread by hand at 16 kg/ha over 2.56 ha centred above cave systems in Trials 1-3 and over 20.7 ha and two caves in Trial 4. Totals of 460, 202 and 95 mice were ear-tagged prior to bait spreading in Trials 1 - 3, respectively, to identify resident mice within the core of each study area. A total of 940 mice were subsequently caught with 100% bait acceptance by ear-tagged mice in all trials. All mice caught in caves were positive for rhodamine-dyed bait, indicating that cave systems are unlikely to be an obstacle for eradication. Our results indicate that mouse eradication could be successfully conducted on Gough Island and that planning for such an operation should proceed in order to remove the key conservation threat to the island's wildlife.

Keywords: House mouse, *Mus musculus*, Tristan albatross, *Diomedea dabbenena,* conservation

INTRODUCTION

House mice (*Mus musculus*) introduced to temperate/sub-Antarctic islands can have serious negative effects on seabirds and other species (Angel and Cooper 2006; Cuthbert and Hilton 2004; Jones *et al.* 2003; Ryan and Cuthbert 2008; Smith *et al.* 2002; Wanless *et al.* 2007). On Gough Island, these effects have resulted in the Tristan albatross (*Diomedea dabbenena*) and Gough bunting (*Rowettia goughensis*) being given a conservation status of Critically Endangered and Atlantic petrel (*Pterodroma incerta*) as Endangered (IUCN 2010). Mice also prey on the chicks of great shearwaters (*Puffinus gravis*) (Wanless *et al.* 2007) and sooty albatrosses (*Phoebetria fusca*) (RSPB unpublished data). Furthermore, many populations of burrowing petrels have decreased dramatically over the last few decades (Ryan 2010). Population modelling for the Tristan albatross and Atlantic petrel suggests that mice are driving these population declines (Cuthbert *et al.* 2003; Cuthbert 2004; Wanless *et al.* 2009).

Given their recorded and potential impacts (Smith *et al.* 2002; Jones *et al.* 2003; Ryan and Cuthbert 2008; Jones and Ryan 2010), strategies for eradicating mice from large islands are needed. At present, when mice are compared with rats on islands, the failure rate of mouse eradication attempts is higher (Howald *et al.* 2007; MacKay *et al.* 2007) and the maximum area from which mice have been successfully eradicated is smaller (710 ha Enderby Island v. 11,300 ha Campbell Island; McClelland and Tyree (2002), Torr (2002)). This means that the outcome of an eradication attempt on 6400 ha Gough Island is uncertain. The feasibility of eradicating mice from Gough Island was recently assessed by Parkes (2008), who concluded that an eradication was technically feasible, but that key questions remained to be answered prior to an operation being undertaken.

To provide confidence to operational managers and potential funders that an eradication operation is likely to succeed, trials have been used to determine the levels of bait acceptance by target species. Typically, these trials utilise non-toxic bait stained with a biomarker dye, with the baits spread at the likely density and time of year as the proposed operation. Such trials were undertaken for rats on Campbell Island (P. McClelland pers. comm.) and Lord Howe Island (I. Wilkinson pers. comm.) and recently at Gough Island (Wanless *et al.* 2008). Following near total bait acceptance in the first two trials, operations on Campbell went ahead and plans for Lord Howe Island are now close to being realised.

On Gough Island, eradication attempts are complicated by large size, mountainous terrain and numerous caves, including lava tubes up to 20 m long (Parkes 2008). The caves are used as breeding sites by hundreds of broad-billed prions (*Pachyptila vittata*) (Cuthbert 2004) and may contain sufficient food to obviate the need for mice to forage outside. Mice could thus fail to encounter bait pellets (Parkes 2008; Wanless *et al.* 2008). If this were the case, some mice may only be killed if caves are targeted specifically – a logistically challenging endeavour given that only a fraction of the island's caves have been identified. Nonetheless, operation managers must be confident that aerially applied bait will be accessible to the mice in caves (Parkes 2008; Wanless *et al.* 2008). Before a full Operational Plan can be completed for a mouse eradication on Gough, the following steps remain: (1) define and test the optimal bait and baiting procedure, (2) determine whether all mice within caves systems will take aerially distributed bait, and (3) conduct bait acceptance trials that replicate eradication conditions in the field.

In this study, we present results of bait trials on Gough Island to determine the susceptibility of mice, including those in caves, to an aerial drop of bait. These trials build on the work of Wanless *et al.* (2008) who found that 3% of mice avoided bait in a trial conducted on Gough in 2006. Confounding effects of the study design may account for these results, but if some mice rejected the bait, the prospects for successful eradication are uncertain (Wanless *et al.* 2008). These authors also found that mice in a cave took surface bait. However, the small number of mice used (11), the small sample of caves (1), and the way bait application differed from aerial spread, limit the conclusions that can be made for the island as a whole.

We undertook further trials above ground and around three separate cave systems. We ear-tagged mice before bait was spread within the core of the first three trials (as on Lord Howe Island and recommended by Parkes (2008)

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and Wanless *et al.* (2008)) and conducted a further trial over a larger area (as on Campbell Island). Our study was thus able to remove the factors that confounded previous trials on Gough Island and provide empirical measures of potential for the success or failure for a mouse eradication attempt.

MATERIALS AND METHODS

Study area

Gough Island (40 °13'S, 9°32'W) is part of the United Kingdom Overseas Territory of Tristan da Cunha, and lies in the central-South Atlantic Ocean some 2600 km from South Africa and 380 km southeast of Tristan da Cunha (Fig. 1). The island is steep and mountainous rising to 910 m above sea level (asl). Annual precipitation is around 3100 mm and higher altitude areas are often shrouded in mist and cloud. Lowland areas are dominated by fern bush vegetation, characterised by relatively tall (up to 3–4 m), island cape myrtle (*Phylica arborea*) trees, dense ferns and sedges, whereas upland areas comprise low-lying wet heath habitat, peat bogs and bare rocks (Wace 1961).

Bait acceptance trials

Movement distances

This part of the study was based on the movements of mice on Gough Island in winter. Eight radio-tagged mice were observed at 160 locations, and 373 live trapped mice were recaptured 1584 times on four 8 x 8 m grids of 100 traps situated in lowland (n=2) and upland (n=2) areas. For mice previously captured in caves, the minimum distance moved was estimated as the distance from the cave-entrance to the trap on the trapping grid.

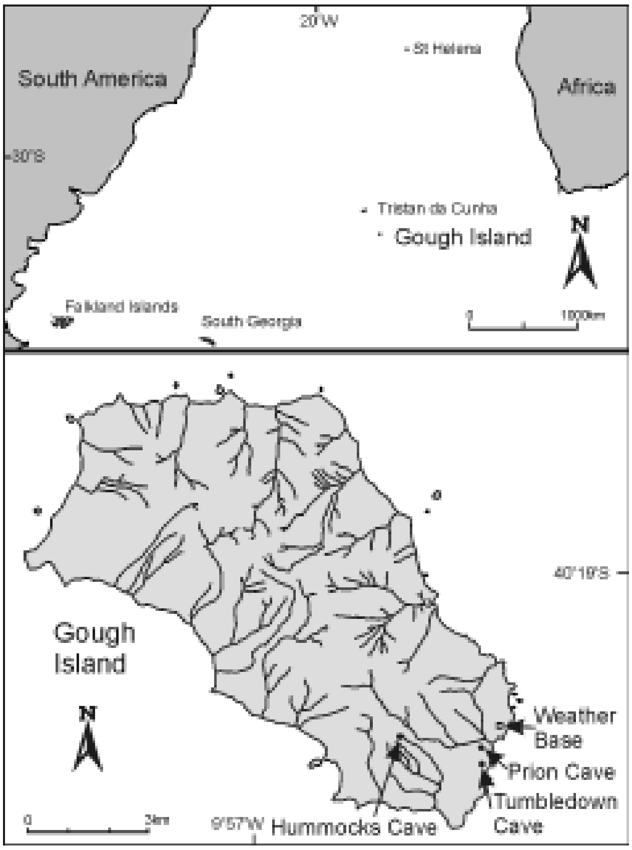

Fig. 1 Gough Island is part of the United Kingdom Overseas Territory of Tristan da Cunha, in the central-South Atlantic Ocean. Trials were undertaken around Prion Cave, Tumbledown Cave and Hummocks Cave.

Susceptibility to baits

Four bait acceptance trials were undertaken, with three in lowland areas (Trials 1, 2 and 4; C. 50 m asl) and one in the uplands (Trial 3; 530 m asl). Trials 1-3 were conducted in winter: mid June (Trial 1), early July (2) and late July (3). Trial 4 was at the onset of spring in late September.

Trials 1-3 were around Prion Cave, Tumbledown Cave and Hummocks Cave respectively (Fig. 1). Mice were caught within caves and on a 72 x 72 m trapping grid outside caves with the cave entrance at its centre. One hundred single catch live-traps were set outside and 3-12 multi-catch live-traps were set within caves for four consecutive nights. All mice captured were fitted with individually numbered ear-tags (Vet Tech Solutions, UK). Bait was then spread over a 2.56 ha area (160 x 160 m), with the cave and trapping grid at its centre and a minimum distance from the outer edge of the baiting to the core trapping-grid (buffer zone) of 44 m.

Mice were not ear-tagged in the core area of Trial 4 as the baited buffer zone was a minimum of 180 m beyond the trap grid and thus well beyond the maximum distance moved by mice entering the grid from outside. The baited area of Trial 4 measured 20.7 ha (ca 397 x 598 m) and overlapped the caves of Trials 1 and 2.

Non-toxic cereal bait pellets (PESTOFF20R, Animal Control Products, New Zealand) with the same formulation as toxic bait were used for the trials. Rhodamine dye was applied to bait on Gough Island, following protocols recommended by the manufacturer. The palatability of baits to rodents is not affected by rhodamine concentrations in the range used to mark bait (Fisher 1999), so the results of these trials should be directly comparable to a toxic bait operation.

In all trials, baits were spread by fieldworkers walking line-abreast along linear transects and spreading bait by hand over a 4-5 m swathe on either side to simulate aerial spread. Bait density was 16 kg/ha over 2.56 ha for Trials 1-3 and 16.9 kg/ha over 20.7 ha for Trial 4. No bait was spread in the caves.

Beginning one day after the baits were spread, mice were kill-trapped for three consecutive nights in Trials 1-3 and four consecutive nights in Trial 4. Two hundred snap traps and 100 live traps were set within the core area (72 x 72 m) of each trial, with 2 snap traps and 1 live trap set at each grid-point. In addition, 3-12 multi-catch live traps and additional snap traps were set in the cave systems.

All mice were checked with an ultraviolet light for the presence of rhodamine at the mouth and anus and within their intestinal tract (Jacob *et al.* 2002). When results were unclear, 6-12 whiskers were collected from each animal, washed in ethanol, and stored for examination under

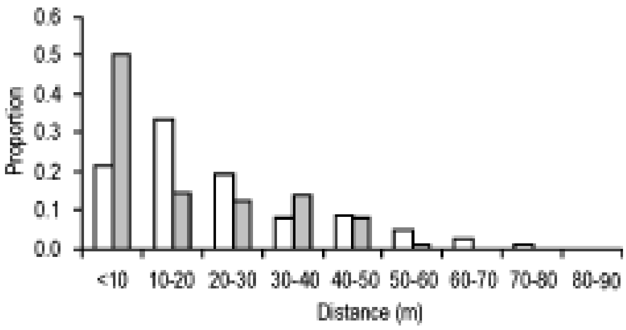

Fig. 2 Frequency distribution of distances moved by mice during the three nights of live-trapping and single night of kill-trapping for trials 1, 2 and 3, for mice captured above ground (unfilled bars) and mice initially caught within caves and subsequently captured above ground (shaded bars).

a microscope and/or hand-lens. Vouchers for positive samples of whiskers were obtained from 20 mice scored positive from their stomach contents. Negative samples were obtained from 20 mice before the baits were spread. Information on sex and reproductive status was collected from all kill-trapped mice.

Potential mouse food resources within caves

If mice in caves were to avoid poison bait outside they needed an alternative source of food. This was most likely to be associated with breeding broad-billed prions within the caves. Monthly checks were conducted at several caves (including those used in Trials 1-4) during the year to record whether birds were breeding and if there was any evidence of predation by mice. Caves were also searched for the presence of invertebrates and other potential food resources.

RESULTS AND DISCUSSION

Movement distances

Over 95% of recorded overnight movements were <40-50 m, with <1% of movements >80 m (R. Cuthbert unpublished data). Mice on the trapping grid most frequently moved 10-20 m (Fig. 2). When mice originally caught within caves are compared with those originally caught above ground, the mice in caves moved shorter distances (Fig. 2). However, this ignores the 10-20 m mice must move within the caves to reach the entrance. Even though 50% of the mice originally from caves were caught < 10 m from the cave entrance and >90% were within 30-40 m of the cave, all mice left the caves when bait was available outside.

Bait trials

Before the baits were spread, 460, 202 and 95 mice were ear-tagged in Trials 1, 2 and 3, respectively. After the baits were spread, 811 mice were captured, with numbers decreasing in sequence from Trials 1 to 3 (Table 1). These declines probably reflected decreasing mouse densities during winter and lower densities of mice in highland areas (Trial 3).

The percentage of mice recaptured also decreased within each trial, with 85%, 41% and 16% over nights 1, 2, and 3 (respectively) in Trial 1 and 83%, 50% and 14% in Trial 2. In Trial 3, few mice were captured on the second and third nights (Table 1), probably as a result of kill-trapping the resident (tagged) mice. In this trial increasing proportions of (non-tagged) mice from the outer zone were captured on nights 2 and 3.

Of the 811 mice examined in Trials 1-3, 810 (99.9%) were positive for rhodamine dye. One untagged mouse caught on night one of Trail 1 tested negative. Of the 368 ear-tagged mice that were re-trapped, all were positive for rhodamine. The dye was clearly visible within the intestines or mouth and anus of all but two mice. Whiskers examined from these two indicated rhodamine on one mouse but no evidence of rhodamine on the second.

Of the mice caught during Trials 1-3, 422 mice were female and 389 male (not significantly different from an equal sex ratio, χ^2=1.26). No females were pregnant and neither sex showed signs of reproductive activity, which reflects the winter trapping period (Jones *et al.* 2003).

Despite increased trapping after the spread of bait for Trial 4, only 116 mice were captured although all of them were positive for rhodamine (Table 2). The small number of mice trapped likely reflected the effects of season and size of the trapping grid. In early spring, mice numbers are at their lowest, and the much larger area baited provided little incentive for peripheral mice to move into the trapping grid.

In the caves, 122 mice were captured during Trial 1 over four nights of live trapping before baits were spread, but only six mice were captured in caves after baits were spread. Similarly, 44 mice were captured during Trial 2 in the cave before baits were spread, but only six were captured in the cave after bait distribution. For Trial 3, six mice were live-trapped in caves before baiting with two re-caught after baits were spread. These results suggest that with abundant food outside caves, most mice previously captured from inside the caves moved out to forage. Furthermore, although both caves in Trials 1 and 2 were within the larger area baited in Trial 4, no mice were caught in the caves despite four nights of trapping. This also suggested that when food was abundant outside, mice moved out of the caves.

During Trials 1-3, 148 mice marked inside caves were recaptured outside, and 14 mice were recaptured inside the caves following bait distribution. All of these mice tested positive for rhodamine.

Table 1 Numbers of house mice trapped on Gough Island over the three consecutive nights of trapping and for the total period of Trials 1-3.Numbers of ear-tagged individuals retrapped above ground from within cave systems are shown in parentheses.

Trial	Night 1			Night 2			Night 3			Total		
	New	Retrap	Total	New	Retrap	Total	New	Retrap	Total	New	Retrap	Total
1	20	118 (3)	138 (3)	79	56 (1)	135 (1)	168	32 (2)	200 (2)	270	203 (6)	473 (6)
2	14	68 (6)	82 (6)	16	16 (0)	32 (0)	147	24 (0)	171 (0)	176	109 (6)	285 (6)
3	9	37 (0)	46 (0)	1	6 (2)	7 (2)	0	0 (0)	0 (0)	10	43 (2)	53 (2)

Table 2 Summary statistics of trapping effort after bait spreading for house mice over the four cave trials and results for presence or absence of rhodamine dye after bait spreading for both ear-tagged and non-tagged mice.

Trial	Nights trapped	Traps set	Mice killed		Tagged		Non-tagged	
			Grid	Cave	Positive	Negative	Positive	Negative
1	3	900	479	6	209	0	269	1
2	3	900	291	6	114	0	177	0
3	2	600	55	2	45	0	10	0
4	4	1200	116	0	-	-	116	0
Total	12	3600	941	14	368	0	572	1

During Trials 1-3, baits were still visible on the ground two days after they were spread and in Trial 4 (in early spring) baits were visible for >10 days. This suggests that baiting densities used in the trial areas were sufficient to provide bait for all mice present.

Potential food resources within caves

Monthly visits indicated that broad-billed prions entered the caves in September, incubated eggs during November-December, reared chicks from December to March, and had departed by April/May. There were few remains of chicks or eggs within caves in winter and no invertebrates were found. In November, some eggs had holes that were nibbled by mice, and in January, February and March, seven prion chicks were found with sign that mice had fed on them. It was not clear whether these were examples of predation or scavenging.

CONCLUSION

Bait trials on Gough were designed to closely mimic the suggested design for an eradication (Parkes 2008) in terms of time of year, bait density and bait formulation. There was 100% bait acceptance in three trials and 99.8% in the fourth, with one mouse negative for bait out of 479 examined. This mouse, which was not captured and ear-tagged in the study grid prior to the spread of bait, may have subsequently moved into the study area. Supporting this inference, all ear-tagged mice resident to the study areas were positive for rhodamine-dyed bait. Moreover, all mice caught within the cave systems before the bait application later tested positive for rhodamine dye, regardless of whether they were re-caught above or below ground. Visits to multiple caves on Gough confirmed conclusions by Wanless *et al.* (2008) that during winter, the absence of breeding birds and other food resources would provide little food for mice.

Our results differ from a previous bait acceptance trial on Gough Island (Wanless *et al.* 2008), where 3% of mice were negative for bait. Combined with relatively high failure rates for mouse eradications, this result has led conservation decision makers in the UK to express concern about the likelihood of success of an eradication operation on Gough. However, with the use of ear-tagged mice, trials over a larger area, and trapping the mice immediately after baits were spread, our study provides greater confidence of a successful result.

Furthermore, given that all four trials on Gough found 100% bait acceptance by resident tagged mice and by non-tagged mice within the larger trial, planning for an operation on Gough Island should now proceed. The final steps in feasibility analyses will now involve evaluating the risk of primary and secondary poisoning to non-target species and captive husbandry trials of potentially vulnerable land birds. Whether there are additional obstacles to eradicating mice from Gough depends on the husbandry trials and the results of attempts to eradicate mice from Coal Island in Fiordland and Rangitoto/Motutapu islands in New Zealand, and Macquarie Island in Australia's sub-Antarctic. If these indicate no fundamental obstacle to removing mice from large islands, the eradication of mice should proceed on Gough Island, a key conservation threat to this World Heritage Site would be removed, and the recovery of Gough's threatened wildlife would become possible.

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Considering native and exotic terrestrial reptiles in island invasive species eradication programmes in the Tropical Pacific

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Abstract Most island restoration projects with reptiles, either as direct beneficiaries of conservation or as indicators of recovery responses, have been on temperate or xeric islands. There have been decades of research, particularly on temperate islands in New Zealand, on the responses of native reptiles to mammal eradications but very few studies in tropical insular systems. Recent increases in restoration projects involving feral mammal eradications in the tropical Pacific have led to several specific challenges related to native and invasive reptiles. This paper reviews these challenges and discusses some potential solutions to them. The first challenge is that the tropical Pacific herpetofauna is still being discovered, described and understood. There is thus incomplete knowledge of how eradication activities may affect these faunas and the potential risks facing critical populations of these species from these eradication actions. The long term benefit of the removal of invasives is beneficial, but the possible short term impacts to small populations on small islands might be significant. The second challenge is that protocols for monitoring the responses of these species are not well documented but are often different from those used in temperate or xeric habitats. Lizard monitoring techniques used in the tropical Pacific are discussed. The third challenge involves invasive reptiles already in the tropical Pacific, some of which could easily spread accidentally through eradication and monitoring operations. The species posing the greatest threats in this respect are reviewed, and recommendations for biosecurity concerning these taxa are made.

Keywords: Invasive reptiles, glue (sticky) traps, mammal eradications, geckos, skinks, iguanas

INTRODUCTION

Most island restoration projects with reptiles, either as direct beneficiaries of conservation or as indicators of recovery responses, have been on temperate or xeric islands (Towns *et al*. 2006). There have been decades of research on the responses of native reptiles to mammal eradications, particularly on temperate islands in New Zealand, but very few studies in tropical insular systems (e.g., Kessler 2002). Most published papers that identify the effects of invasive mammals on tropical Pacific reptiles focus on ungulates or carnivores (e.g., Gorman 1975; Pernetta and Watling 1978; Kirkpatrick and Rauzon 1986; Case and Bolger 1991; Harlow *et al*. 2007), and there is little information on the effects of rodents (Case *et al*. 1991; Towns *et al*. 2006). Recent increases in restoration projects involving the eradication of introduced mammals in the tropical Pacific have led to several specific challenges related to native and invasive reptiles. I review these challenges and suggest potential solutions to some of them.

The first challenge is that the reptiles of the tropical Pacific are still being discovered, described, and understood. This leads to incomplete knowledge of how eradication programmes may affect these faunas and the nature of potential risks to critical populations. It also impedes our ability to prioritize restoration efforts for reptiles, since the factors impacting species with reduced population numbers are not often known (McCoid *et al*. 1995; Fisher and Ineich in press).

The second challenge is that methods for monitoring the responses of these reptile species to specific management actions are not well documented and are often different from those used in temperate or xeric habitats (Gillespie *et al*. 2005; Ribeiro-Junior *et al*. 2008). Reptile survey techniques being used on Palmyra Atoll, Line Islands, and the Aleipata Islands, Samoa (Fig. 1), to measure responses to rat eradication projects are reviewed below but there are many other techniques and protocols that can be used. Documentation and standardisation of procedures and accuracy in species identification are the most important long-term elements in establishing effective management programs.

The last challenge is that there are many invasive species of reptiles already in the tropical Pacific, particularly on Hawai'i and Guam (McKeown 1996; Kraus 2009) and some could easily be spread inadvertently through management actions, especially if such trips are the only visits to remote island locations. The species most likely to spread in the tropical Pacific are reviewed below and recommendations for biosecurity for these taxa are discussed.

REPTILE KNOWLEDGE GAPS IN THE PACIFIC

Research on reptile diversity in the Pacific lags behind the more conspicuous groups such as birds. Although the herpetofaunas of most archipelagos have generally been well documented (e.g., Bauer and Henle 1994; Gill 1993, 1998; Gill and Rinke 1990; McCoy 2006; McKeown 1996; Morrison 2003; Zug 1991), faunal lists for many individual islands do not exist. Many newly recognised species remain undescribed including geckos, skinks, and blind snakes (Bruna *et al*. 1996; Radtkey *et al*. 1995 Fisher 1997; Wiles 2004; Watling *et al*. 2010; Buden and Panuel 2010; Wynn *et al*. in review), and there are others described during the past 25 years that are still known from single localities and/or few specimens (Zug 1985; Ota *et al*. 1995; Zug and Ineich 1995; Zug *et al*. 2003; Buden 2007; Ineich 2008,

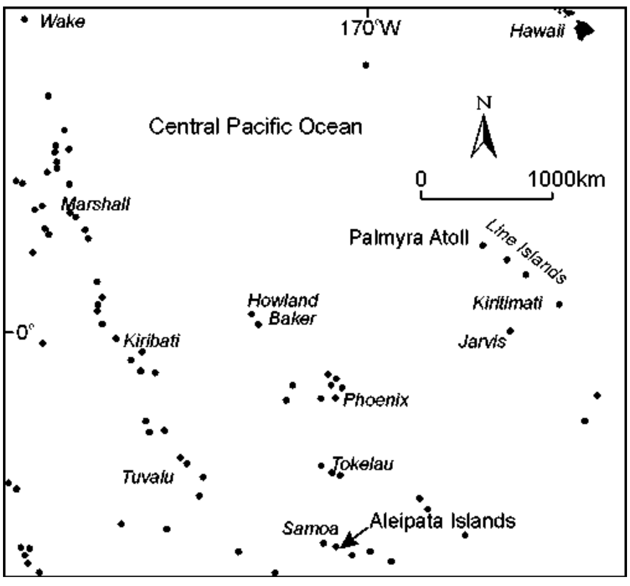

Fig. 1 Location of Palmyra Atoll and Aleipata Islands in the Pacific Ocean.

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2009). In addition, some taxa that are known from only one or a few individuals and are presumed extinct could potentially be rediscovered (Ineich and Zug 1997).

Fossil deposits show that reptile faunas were once more diverse on several island groups before the arrival of people and invasive species (Pregill 1993; Crombie and Pregill 1999; Steadman 2006; Pregill and Steadman 2009). Some of these taxa may persist on small refuge islands as this has been shown to be a pattern elsewhere in the Pacific where species are now lacking from the main islands (e.g., Pernetta and Watling 1978; Perry *et al*. 1998; Steadman and Pregill 2004; Towns and Daugherty 1994). We currently know of three new species that appear to have relictual distributions due to the extirpation of insular populations prior to discovery (Pregill and Steadman 2009; Watling *et al*. 2010; Buden and Panuel 2010; Wynn *et al*. In review).

A particular problem in the Pacific is that different reptile species can be superficially similar in appearance. For example, the island groups of the central and south Pacific often have between two and four species of small *Emoia* ground skinks, all of which are striped (Fig. 2), similar-looking species of striped *Lipinia* skinks on the ground or in trees, and a striped *Cryptoblepharus* shore skink. Because supporting museum collections for many areas are often poor or incomplete, any records that are based solely on identification by sight – without capture and study – can lead to errors.

Little is known about the impacts of rodenticides or other toxicants on reptiles (Hoare and Hare 2006). Biomarker studies being carried out on several tropical islands may indicate how the toxicants move through the food webs (Wegmann *et al*. 2008). Fossorial species, such as blind snakes, might be indirectly affected by rodent bait campaigns that introduce toxicants into the soil, either by direct exposure through the soil or secondarily by consumption of contaminated ant pupae and other foods (Ogilvie *et al*. 1997). On Indian Ocean islands, skinks have been directly observed eating rain-softened bait pellets, although no direct mortality was observed (Merton *et al*. 2002); identifying a potential direct risk of poisoning to similar rare species on Pacific Islands (i.e. *Tachygia*, *Emoia slevini*; Ineich and Zug 1997).

Often small islands retain bird populations that are identified as key beneficiaries from eradication programmes. However, a thorough evaluation for reptiles on these islands prior to implementation of any feral mammal eradication is warranted and should be required. Small islands often have relict populations of rare or threatened reptiles and/or have high value for reptile conservation. Furthermore, reptiles may be affected directly or indirectly by eradication programmes.

USE OF APPROPRIATE MONITORING PROTOCOLS

Techniques employed for monitoring reptiles in tropical environments include nocturnal and diurnal visual encounter surveys, pit-fall trap, sticky-trap, road search, and removal plots (Rodda *et al*. 2001; Gillespie *et al*. 2005; Ribeiro-Junior *et al*. 2008). Some techniques commonly used on temperate or xeric islands, such as pit-fall traps, do not work well in the tropical Pacific due to differences in habitat and the biology of the target species. For example, the species present on many islands are often predominantly arboreal skinks and geckos. A combination

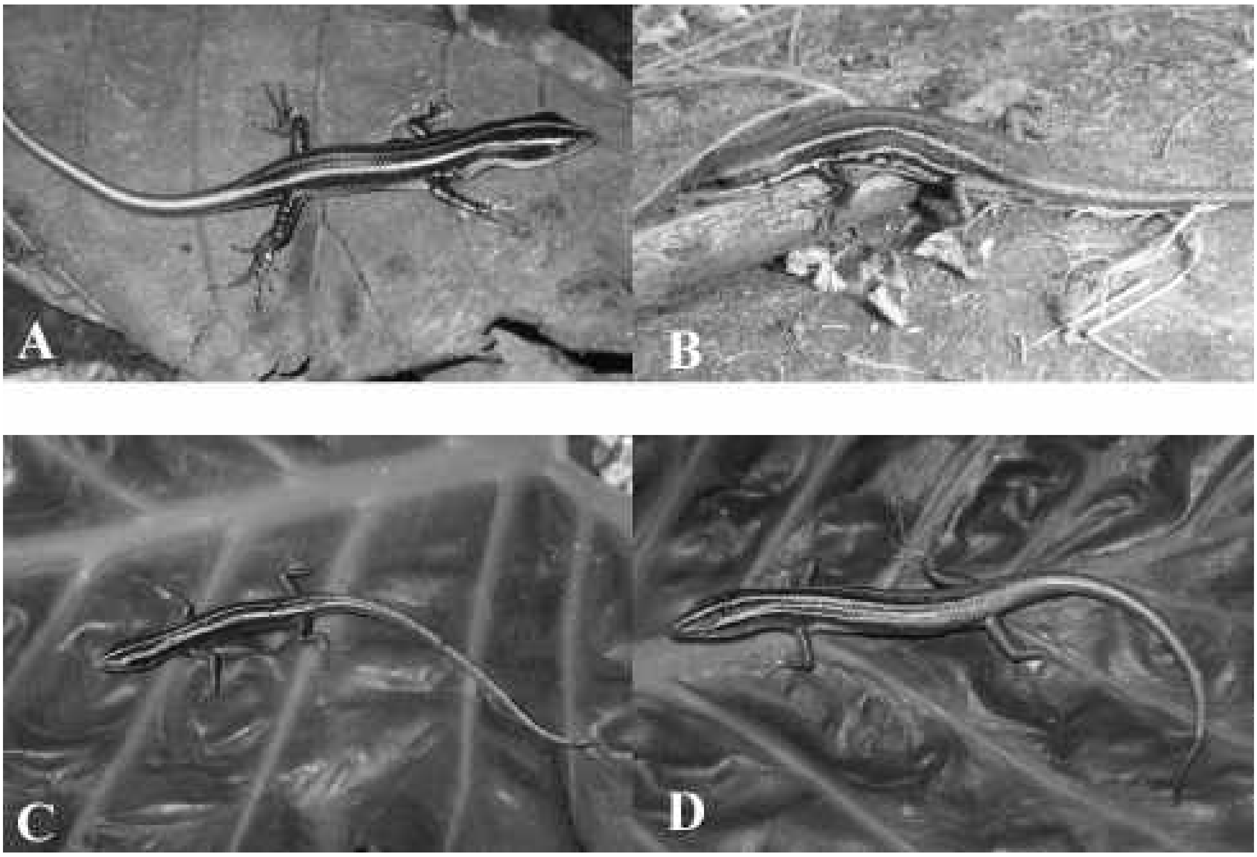

Fig. 2 Superficial similarity in appearance can cause difficulties for visual identification as illustrated by these four widespread striped skinks in the genus *Emoia* from the Pacific Basin. A. *Emoia caeruleocauda* is widespread in Micronesia, Solomon Islands, Vanuatu, and one island in Fiji. It is not endemic to the Pacific Basin but occurs in New Guinea and Indonesia also. B. *Emoia jakati* is introduced into the Solomon Islands and apparently much of Micronesia. It is native to New Guinea (Photo courtesy Don Buden). C. *Emoia impar* is endemic to the Pacific Basin and may represent several different cryptic species. This species has been extirpated from Hawai'i. D. *Emoia cyanura* is endemic to the Pacific Basin and may represent several different cryptic species. This species was accidentally introduced to Hawai'i in the 1970s and later extirpated.

of several techniques should give more information and help identify conservation targets for long-term restoration projects. Such an approach should also reveal targets for monitoring the responses of the eradications. A key factor is to ensure that the people monitoring reptile populations have the appropriate training and equipment. The priority here is to be able to accurately identify the species to be counted and measured. Some species are widespread and easy to identify, whereas others are part of cryptic species complexes, or are similar in appearance to invasive species. The lack of good regional field guides (exceptions are Morrison 2003; McCoy 2006) is part of this problem, as is the dearth of reptile specialists in the field through much of this region.

When used in combination and under similar environmental conditions, the following three techniques will provide repeatable relative measures of the contributions of species in reptile communities. These methods will provide baseline data on reptile communities and later measure the response to eradication actions. The same methods can also be used for biosecurity screening as they will detect most of the invasive reptile species in the Pacific region.

1. Visual Encounter Surveys – Daytime: These use transects traversed on foot across various habitats (Case and Bolger 1991), preferably during fine weather; i.e. not on overcast or rainy days. Each transect should cover a different habitat type or sampling stratum, with any reptiles observed along these transects recorded along with transect length and sampling duration. There are many ways to do these surveys, and they can be quantified either by fixed amount of time, fixed distance, or fixed route, or combinations of these. Whatever is done needs to be well documented so it can be repeated in the future. Validation that the observer can identify the various species present is required before using this or any direct-count technique. This technique works only for day active species, including diurnal skinks, monitor lizards, iguanids (particularly invasive *Anolis* and *Iguana*), and diurnal geckos (e.g., *Phelsuma*). Changes in vegetation cover after an eradication might make repeatability of these surveys difficult, especially if the vegetation becomes too thick to detect reptiles.

2. Visual Encounter Surveys – Night time: These use the same methods as daytime Visual Encounter transects, but can also include village buildings or other structures (Case *et al*. 1994). Bright headlamps or flashlights should be used to detect animals; some observers use a combination of flashlight with binoculars to increase focus on distant observations. As with daylight surveys, appropriate environmental conditions are preferred for comparing across nights and rainfall should be avoided. This technique works best for nocturnal geckos and boids, some invasive taxa (e.g., rats), and also coconut crabs and other species of interest (Harlow *et al*. 2007). It can also be effective for some diurnal species that roost in the canopy such as *Brachylophus* iguanas.

3. Glue (or Sticky) Trap Transects: Although there is often animal welfare concern over the use of this technique, proper application avoids or greatly reduces mortality of the trapped individuals (Ribeiro-Junior *et al*. 2006). Glue traps are generally cheap, easy to deploy, and work well in situations where the vegetation or other features (rocks/trees) are thick and animals are difficult to find. I have used traps set every 10-25 metres in transects that are 100 (or 250 m) long, the distances between sets and the length of the transects depending on the nature of the study. Each trap site consists of three sticky traps: one on the ground, one on a log, and one on a tree. The traps can be set and checked every 15 minutes for about 2 hours. The strength of adhesion varies by trap brand and weather conditions. Traps may be ineffective within a few hours if hit by direct sun, which should be avoided anyway as it will kill any trapped animals. Other traps last for days, even during rain, although cardboard backed ones will fall apart if they get too wet. Glue traps can also be set in the late afternoon and left overnight to capture geckos and rats, although this often leads to higher mortality due to ant and land crab predation. Trapped animals can be removed using a thin coat of vegetable oil on the operator’s fingers and then slowly peeling the animals off of the trap. Lizards can be toe clipped or marked with a temporary mark (felt pen, paint, etc.) to assess future recaptures; these same techniques can also be used for visual transects if animals are captured. Additional data such as invertebrate samples can be collected from the sticky traps if they are wrapped in plastic-film and frozen for later analysis. Each island should have 2–5 transects depending on island size and study questions. This can prove a useful way to confirm day or night time visual identifications along transect lines.

INVASIVE REPTILE SPECIES

Appropriately designed surveys may also reveal the presence of some of the following invasive reptile species. Many of these species have a high risk of spread throughout the tropical Pacific and potentially devastating effects on native and endemic species.

Geckos: The rapid invasion of the Asian house gecko (*Hemidactylus frenatus*) across the Pacific has been well documented (e.g., Case *et al*. 1994; Fisher 1997) and its impacts on endemic geckos in the Indian Ocean were described by Cole *et al*. (2005). More recently the spread of the gold-dust day gecko (*Phelsuma laticauda*) south from Hawai’i has become a concern (Ota and Ineich 2006) after it rapidly invaded the Hawaiian Islands from introductions via the pet trade (McKeown 1996). In Hawai’i, the species uses the night-light niche, which could make it a predator or competitor of native geckos as it spreads across the Pacific (Perry and Fisher 2006; Seifan *et al*. 2010). Currently there are many native and invasive geckos in southeast Asia and Hawai’i that could easily spread into the Pacific and impact the natural gecko communities. Many geckos are extremely successful invaders, which spread as adults through shipping, but also because some species with adhesive hard-shelled eggs deposit them under the lips of buckets, in building materials, and other inconspicuous locations. Such species also spread rapidly once they invade new usable habitat. The potentially invasive gecko species and the sites they have invaded are listed by Kraus (2009). Islands that currently lack certain invasive reptiles, such as the Asian house gecko, should be identified. Maintaining them free of such species will require much vigilance and outreach to local communities.

Skinks: Little is known about the impact of skinks, but the curious skink (*Carlia ailanpalai*) is rapidly spreading through Micronesia, replacing native ground skinks (Buden 2009). Two additional species, *C. mysi* and *C. tutela*, have invaded Bougainville and Palau (respectively), which indicates that the genus contains many highly invasive species (Crombie and Pregill 1999; Zug 2004; McCoy 2006). Other skinks such as *Emoia jakati*, *Lampropholis delicata,* and *Lamprolepis smaragdina* have been present in the Pacific for longer although their impacts are not well studied (Baker 1979; Perry and Buden 1999; McCoy 2006; Kraus 2009; Fisher and Ineich 2011; Fisher and Richmond unpub. data). Continued vigilance is necessary to ensure that these species do not spread further. Recently, *Lampropholis delicata*, which is invasive in New Zealand, was intercepted through biosecurity screening in a shipment of timber to Raoul Island, Kermadec Islands, which lack indigenous terrestrial reptiles (Phil Bell pers. comm.). The species has apparently become recently established on Lord Howe Island (Kraus 2009).

Iguanids (and *Polychrotids)*: Several species of anoles or American chameleons (Poychrotidae: *Anolis*) are now in Hawai'i, Commonwealth of the Northern Mariana Islands and Guam (McKeown 1996; Kraus 2009). Studies of *Anolis carolinensis* in the Ogasawara Islands (Suzuki and Nagoshi, 1999) indicate that anoles could compete with the endemic skinks of the Pacific Basin. Additionally, *Anolis sagrei* in Hawai'i apparently uses the same habitat as the native *Cryptoblepharus poecilopleurus* (Fisher pers. obs.) although the effects of this need further study. Additionally, green iguanas (Iguanidae: *Iguana iguana*), which have been in Hawai'i at least since the 1950s (McKeown 1996), were introduced to Fiji early this century and now threaten endangered Fijian iguanas (*Brachylophus* spp.; Naikatini *et al*. 2009; Thomas *et al.* 2011). Restricting the spread of green iguanas in Fiji is a major concern. The potential impact of the species on the vegetation community is unknown since the invasion is just now irrupting.

Chameleons: Jackson's chameleon (*Chamaeleo jacksonii*) was introduced into the Hawaiian islands in the early 1970s and is now widespread (McKeown 1996). The species had not spread beyond Hawai'i until recently, when the chameleons were reported from the Marshall Islands (Vander Velde 2003). In Hawai'i, the species preys on endemic and endangered invertebrates, which adds to the pressure on these species (Holland *et al*. 2010).

Snakes: There is an extensive literature on the brown tree snake (*Boiga irregularis*) and the threats posed by the species are well known (Rodda and Savidge 2007). Other snakes such as the wolf snake (*Lycodon aulicus*) also appear to be capable invaders and could threaten the endemic Pacific fauna if it spreads from southeast Asia (Buden *et al*. 2001; Cheke and Hume 2008; Kraus 2009). The flower pot snake (*Ramphotyphlops braminus*) continues to spread throughout the Pacific Basin although its impacts are not well known (Kraus 2009). With the recent discovery of new endemic species of blind snakes (*Ramphotyphlops spp*.) within the oceanic Pacific, concern over confusion between indigenous and invasive species increases and other endemic species might go unnoticed and unprotected (Buden and Panuel 2010; Watling *et al*. 2010; Wynn *et al*. in review). Competition between native blind snakes and the invasive flower pot snake might become a concern as the latter species continues to spread.

These invasive reptile species raise biosecurity issues that must be taken very seriously, especially since conservation actions, including eradication efforts, could be a mechanism for their spread. Training tools and protocols for cleansing of equipment and supplies between islands should be developed and rigorously implemented to ensure that restoration projects do not spread unwanted aliens. Geckos pose the greatest threat through their accidental spread with the movement of materials used for remote island restoration activities. Boats are a particular risk pathway for some of these species and require careful planning to minimise this threat when visiting and working on remote islands.

CONCLUSION

Reptiles should be considered an important component of adaptive management projects in the Pacific because there are often endemic or relict populations on remote islands. Because knowledge of these species is often poor, experts should be consulted to ensure that these management actions have positive rather than negative impacts on native species. This is vital, especially in light of the number of highly-localised, poorly-understood endemic species distributed intermittently across the Pacific Basin. Capacity building through species identification courses and the development of better, more exhaustive field guides should be conducted with those who plan to monitor reptile responses to these management activities.

Understanding and managing the biosecurity risks associated with conducting fieldwork at remote sites is vital to ensure that restoration activities do not further the spread of invasive species, such as the Asian house gecko or gold-dust day gecko (Hathaway and Fisher 2010).

Lastly, if priority reptile areas for conservation in the Pacific were mapped, management activities that would benefit multiple taxa could be identified, thereby adding the recovery of reptiles to birds and invertebrates as restoration targets.

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Invasive alien species on European islands: eradications and priorities for future work

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ABSTRACT A high proportion of European endemic species occur in island ecosystems, and many threatened species are affected by invasive alien species. Tackling biological invasions on European islands is therefore crucial for protection of regional biological diversity and, in many cases, for the well-being of local human communities. Europe is one of the richest regions of the globe, but despite its formal commitment to halt the regional loss of biodiversity by 2010, the level of action to prevent, eradicate or control invasive alien species on islands has been so far very scant. In order to provide an updated list of attempted eradications and tools to support a more efficient decision making, a database on invasive species on the islands of Europe has been implemented. It contains information on eradication programmes, the presence of alien species, and native species directly impacted by these. The scope of the database extends over biogeographic borders of the region and covers the outermost territories of Europe. Data have been collected by reviewing scientific literature, unpublished data provided by experts, and reports produced by signatory countries of the Bern Convention. Data on islands have been acquired through cooperation with the Global Islands Database (GID). In Europe, 224 eradication programmes have been carried out on 170 islands, of these eradications 86% have been successfully completed, mostly targeting rats (68%). We discuss options for future work, including prioritisation of actions based on an analysis of island data, threatened species, and key invasives.

Keywords: Europe, management, invasions, prioritisation

INTRODUCTION

European islands host an important component of the region's biodiversity, including a large number of endemics. For example, almost 12% of the flora of Corsica, 10% of the flora in Crete, and 7% of Cyprus are endemic. In the Canary Islands up to 70% of some animal taxa, such as Coleoptera, are endemic (Orueta 2009). The rich biodiversity of European islands is severely threatened by several factors including invasive exotic species. Tackling the impact of invasives in these ecosystems is crucial to reverse the loss of regional biological diversity. The European Strategy on Invasive Alien Species (Genovesi and Shine 2004) schedules special measures for isolated ecosystems to prevent or minimise adverse impacts due to biological invasions. Despite the need to address invasions for protecting the regional biodiversity, the level of action to prevent, eradicate or control invasive alien species on islands in Europe has been scant when compared to other areas of the world. A review by Genovesi (2005) highlighted that few eradications have been successfully completed in Europe, mostly on small islands, and that no invertebrate, plant or marine organisms had been removed. Several reasons were mentioned to explain this limited action, including the lack of adequate legal tools, the scarcity of specific financial resources, and the lack of concern, awareness and public support for these kind of actions.

Following the review by Genovesi, European institutions have adopted several formal commitments to address biological invasions. The Communication on Biodiversity (2006; http://ec.europa.eu/development/icenter/repository/com2006_0216en01_en.pdf), listed invasive alien species as a key priority area of the European Union Action Plan, starting from 2010. A more recent Communication on Biodiversity (November 2008; http://ec.europa.eu/environment/nature/invasivealien/docs/1_EN_ACT_part1_v6.pdf) reaffirmed the need and urgency to develop a European policy on biological invasions. In addition, the European Union Council (June 25th 2009) stressed the urgent need for a strategy on invasive alien species in the European Union, based on the Convention on Biological Diversity (CBD) guiding principles and the document by Genovesi and Shine (2004). Along with these decisions, the European Commission has provided significant financial support to actions aimed at tackling invasive species. In the period 2004-2006 the average annual budget spent for invasive species issues by the European Union LIFE program, and the EU's Framework Programme for Research and Technological Development has exceeded €18 M, in several cases used to carry out eradication programs (Scalera 2009).

To evaluate whether or not the increased political interest in invasions – as well as the improved technical ability to manage invasive species – has led to an increase in the number and complexity of eradications carried on European islands, we provide in this paper an inventory of such programmes. We explore prioritising future actions based on identification of islands, areas and species where funding and efforts should be concentrated. In this context we discuss how available information on the presence of native species threatened by invasives, and invaders with most impact, can be analysed.

MATERIALS AND METHODS

For the purposes of this study, an information system for invasive alien species (IAS) on the islands of Europe has been implemented. This is based on a relational database containing information on 1) geographical parameters of islands (region, area in hectares, geographical coordinates); 2) presence of detrimental IAS; 3) presence of native species directly affected by IAS; and 4) eradication programmes.

The reference list of the most detrimental IAS is based on the DAISIE list "100 of the worst" (DAISIE 2009), the presence of these species on European islands, and on the results of an earlier review paper (Genovesi 2005).

The native species directly affected by IAS have been selected through searches of the Global Invasive Species Database (GISD), the Species Information System of IUCN, and available literature (e.g., Ruffino *et al*. 2009; Banks *et al*. 2008; Bonesi and Palazon 2007).

Data have been collected by reviewing scientific and grey literature (e.g., Howald *et al*. 2007; Nogales *et al.* 2004; Campbell and Donlan 2005) and through a specific questionnaire produced and circulated among key experts. Data on islands have been primarily collected through

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cooperation with the Global Islands Database (GID), which contain information on many different aspects of the world's islands, including biological, social, and economic data.

The scope of the inventory covered in this paper extends over the biogeographic borders of Europe and includes the overseas territories of European countries. The review covers data on all taxa of invasive species, from vertebrates and invertebrates, to plants, but excluding marine aquatic species.

RESULTS

Geographical data on more than 50,000 European islands have been collected, mostly based on information stored in the GID. More detailed information has been gathered for a subset of 197 European islands, where eradication programmes have been carried out.

From the data search, it appears that information on presence/absence of key IAS and native species are rarely available, and in general very scattered. Furthermore, very little information is available at the geographical scale required for prioritisation of eradications. We therefore concluded that at the present time it is not possible to carry out a pilot multi-species prioritisation exercise at the scale we considered.

We recorded a total of 224 eradication programmes reported in (Appendix 1). These have been, or are being, carried out on 170 islands, belonging to 12 different European countries.

Most of the documented eradications have been on islands of the North Atlantic Ocean (n=50) and the Mediterranean Sea (n=45). At present, 11 eradication programmes are in the course of implementation, while a further 16 are completed (but have still to be confirmed). In 17 cases it was not possible to obtain the results of the eradication campaigns (Table 1).

Of the total number of eradication campaigns considered in the present review, final results have been reported for 180 cases; of these projects, 86% are reported as successfully completed, and 14% as unsuccessful. Since successes are in general more likely to be reported than failures, it is possible that the success rate is biased. In three cases (Tuscan Archipelago, Italy) a re-invasion of rats (*Rattus rattus*) has been recorded during a survey carried out some years after the end of a successful eradication (N. Baccetti pers. comm.). The reason is probably the very limited distance (< 500 m) recorded between the islets and the main island, Isola d'Elba, where the species is already present.

The size of the islands where eradications have been attempted ranges from 0.10 ha (Folaccheda, Mediterranean Sea) to 925,100ha (Cyprus, where there was an attempt to eradicate red palm weevil (*Rhynchophorus ferrugineus*)).

Table 1 Overall summary of the status of reported eradications on European islands.

Eradication status	n. eradications
successful	154
unsuccessful	21
uncompleted	5
being confirmed	16
on going	11
unknown	17
Total	224

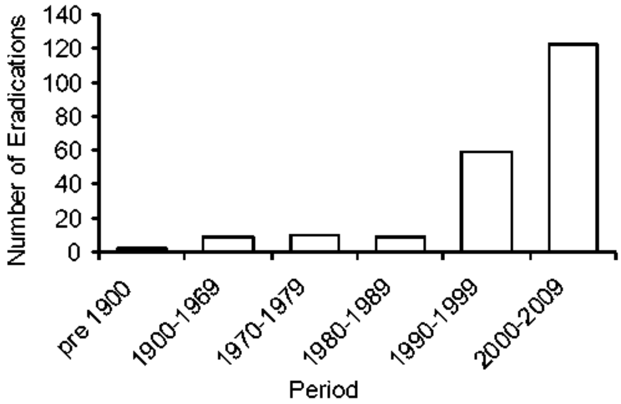

Fig. 1 Frequency of successful eradications; since 1970 reported per decade.

However, the median size of islands where a successful eradication has been reported (n=137) is 17 ha (Q_1=4.0 ha – Q_3 = 288.5 ha), while the median area of islands where an eradication has failed (n=25) is 60 ha (Q_1 = 6.5 ha – Q_3 = 1015 ha). The majority of islands (63%) where a successful eradication has been reported are below 100 ha; three islands where the common myna (*Acridotheres tristis)* has been eradicated are >150,000 ha.

In the last decade, the number of projects carried on European islands has rapidly increased with 58% of successful eradications completed in the 2000-2009 period (Fig. 1).

Thirty five species have been targeted by eradication campaigns, 19 of which are vertebrates, three invertebrates and 12 plants. Rodents account for 66% (n= 137) of all vertebrate eradications, and carnivores and ungulates combined for 23% (n=52) of the total number of projects. Rats (*Rattus* spp.) are the most common target (n=127, 57%), followed by goats (*Capra hircus)* (n=21, 11%).

For 26 eradications (13%) it was possible to gather information on costs. For these cases, the cost ranged from €200 spent for the eradication of ruddy ducks (*Oxyura jamaicensis*) in the Balearic Archipelago (Spain) to €2,247,951 spent so far to eradicate American mink (*Neovison vison*) from the islands of Lewis and Harris in the Outer Hebrides (UK). From the scarce available information it appears that costs can vary much even when the same species is targeted. For example the cost of rodent eradication programmes (n=9) ranges from €321 to €400,000. It was not possible to test for any correlation between costs and eradication area, because of the inaccurate area measurements reported for most programmes.

Regarding the removal techniques, plant eradications have usually been done by mechanical hand removal and animal eradications have been most commonly carried out with poisons, either alone or associated with other removal methods (n=152, 79%). The use of combined techniques was more common in eradications of rats, mice, cats and rabbits. Several successful eradication campaigns (n= 38, 25%) have been carried on by applying several techniques, but this percentage varies widely among target species and in relation to the geographic location of the project. For example, all the eradications of *Rattus exulans* on islands of European overseas territories (n=24) have been conducted using poisons, while for the other two species of rat multiple techniques have been used (n=102, 28%).

DISCUSSION

Information on European islands remains scarce and mostly scattered. No inventory of islands was available before the establishment of the GID (Orueta 2009). Data on the presence of invasive species are not organised, and often available only at the island or archipelago scale. And studies on the impacts caused by invasive alien species to native species are still very limited. No overall information is available for native species on European islands, and very little data have been published on those invasive species with the most devastating impacts. We believe that, for prioritising action to tackle the most impacting invasives, it is necessary to significantly improve the level and scale of information on the presence of threatened species – including on small islands – and of key invasives.

On the other hand, information on attempted eradications is becoming increasingly more accessible and the list of eradication programmes presented in this paper is more comprehensive than previous reviews have reported. A comparison of the data collected for the present study with those reported by Genovesi (2005), confirms the constant increase in the implementation of this management tool in Europe. The range of taxonomic groups targeted by eradications is very wide, and is comparable with the species targeted in other regions of the world (see Genovesi 2005 for a tentative comparison). However, the area of islands where eradications have been attempted in Europe remains quite small. This partly reflects the presence on many European islands of native or endemic species, which imposes restrictions on the removal methods that can be used. The small range of treated islands is also due to the limited awareness of the problems caused by IAS in Europe, and the subsequent limited public support for eradications.

One consequence of the limited awareness of invasions is the often inadequate legal frameworks on this issue. Several toxins have been (or are being) banned, and no derogation procedure has been established for the controlled use of such substances in eradication programmes. Several countries have very strict legislation protecting domestic species, that do not allow the effective management of species such as the domestic cat or dog. It is interesting to note that many complex and technically challenging eradications have been carried on in European overseas territories, located in regions where eradications are less controversial than in Europe.

CONCLUSIONS

Eradication is a crucial tool to mitigate the impacts of IAS and to preserve global biodiversity (Genovesi 2011). The establishment of eradication inventories is important for improving understanding of the technical parameters of this management option, and monitoring the level of action in this regard.

From the information collected for the present study, Europe has increased efforts to combat invasions through eradication campaigns; however, eradications are generally less technically complex and challenging than similar projects attempted in other parts of the world.

In order to improve and strengthen European action on invasive species, it is crucial that any future European policy on invasions has specific provisions on eradications, supporting the realisation of such programmes, addressing the legal obstacles, and providing specific funding devoted to eradications.

Considering the huge number of islands present in Europe, and the fact that in the European system most projects are funded with public funds (e.g., EC, national), particular importance should be placed on establishment of a transparent, science-based prioritisation of programmes. In this regard, the results of this assessment confirm the potential efficacy of an integrated data analysis of islands, native species, and key invasives in order to identify islands, areas and invasive species on which funding and efforts should be concentrated.

Considering the differences in species composition in the different geographic contexts of Europe, any prioritisation work would be more feasible at the regional scale rather than at the continental scale. To allow action prioritisation, it would be useful to develop a list of the invasive species with greatest impact in different European regions (such as Mediterranean, Atlantic, tropical overseas territories, and subantarctic overseas territories).

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Appendix 1 Eradications of alien species carried out on European islands.

Region	Country	Name of island	Invasive species	Erad year	Eradication status	Methods	Ref code
Carribean Sea	FRA	Burgaux	*Rattus rattus*	2002	successful	T, P	17
Carribean Sea	FRA	Fajou	*Mus musculus*	2001	successful	T, P	17
Carribean Sea	FRA	Fajou	*Rattus rattus*	2002	unsuccessful	T, P	17
Carribean Sea	FRA	Fajou	*Oryctolagus cuniculus*	1995	successful	S, P	6,7,22
Carribean Sea	FRA	Fajou	*Herpestes auropunctatus*	2001	successful	T	22,24,32
Carribean Sea	FRA	Hardy	*Rattus rattus*	2002	successful	T, P	17
Carribean Sea	FRA	Percé	*Rattus rattus*	1999	successful	T, P	17
Carribean Sea	FRA	Poirier	*Rattus rattus*	2002	successful	T, P	17
Carribean Sea	NED	Klein Curacao	*Capra hircus*	1996	successful	T	4
Carribean Sea	UK	Bay Cay	*Rattus rattus*	2002	successful	P	17
Carribean Sea	UK	Grand Cayman	*Myopsitta monachus*		on going		3,20
Carribean Sea	UK	Guana	*Capra hircus*	1991	successful	S	4
Carribean Sea	UK	Little Cayman	*Felis catus*		on going	T	3,20
Carribean Sea	UK	Long Cay	*Felis catus*	1999	unknown	P	27
Carribean Sea	UK	Low Cay	*Rattus rattus*	2000	successful	P	17
Carribean Sea	UK	Nonsuch	*Rattus norvegicus*	1985	successful	P	17
Carribean Sea	UK	Nonsuch	*Rattus rattus*	1985	successful	P	17
Carribean Sea	UK	Nonsuch	*Rattus rattus*	2005	successful	P	17
Carribean Sea	UK	Pusey Cay	*Rattus rattus*	2002	successful	P	17
Carribean Sea	UK	Sandy Cay (White Cay)	*Rattus rattus*	2002	successful	P	17
Carribean Sea	UK	Sim Cay	*Rattus rattus*	2002	successful	P	17
Carribean Sea	UK	White Cay (Sandy Cay)	*Mus musculus*	1998	successful	P	17
Carribean Sea	UK	White Cay (Sandy Cay)	*Rattus rattus*	1998	successful	P	17
Carribean Sea	UK	William Dean Cay	*Rattus rattus*	2002	successful	P	17
Indian Ocean	FRA	Amsterdam	*Capra hircus*	1957	successful		4
Indian Ocean	FRA	Australia	*Rattus rattus*	2004	unknown	P	23
Indian Ocean	FRA	Australia	*Mus musculus*	2004	unknown	P	23
Indian Ocean	FRA	Grande Terre	*Oryctolagus cuniculus*	1956	unsuccessful		23
Indian Ocean	FRA	Grande Terre	*Felis catus*	1977	unsuccessful	S	31
Indian Ocean	FRA	Île aux Cochons	*Oryctolagus cuniculus*	1997	successful	P	6,7,22
Indian Ocean	FRA	Île aux Moules	*Rattus rattus*	2005	unknown	P	23
Indian Ocean	FRA	Île du Château	*Rattus rattus*	2002	unknown	P	17
Indian Ocean	FRA	Île du Château	*Mus musculus*	2001	unknown	P	17
Indian Ocean	FRA	Île Guillou	*Felis catus*	1995	successful	S	27
Indian Ocean	FRA	Île Haute	*Ovis aries*	2009	successful	S	23
Indian Ocean	FRA	Île Verte	*Oryctolagus cuniculus*	1992	successful	S, P	6,7,22
Indian Ocean	FRA	Saint-Paul	*Rattus rattus*	1996	successful	P	17
Indian Ocean	FRA	Saint-Paul	*Oryctolagus cuniculus*	1997	successful	P	22
Indian Ocean	FRA	Saint-Paul	*Mus musculus*	1997	unsuccessful	P	23
Indian Ocean	FRA	Saint-Paul	*Capra hircus*	1874	successful		4
Macaronesia	POR	Deserta Grande	*Oryctolagus cuniculus*	1998	successful	S, P	14
Macaronesia	POR	Deserta Grande	*Capra hircus*		uncompleted	S, T	29
Macaronesia	POR	Deserta Grande	*Felis Catus*	1984	successful		27
Macaronesia	POR	Praia islet	*Oryctolagus cuniculus*	1997	successful		14
Macaronesia	POR	Selvagem Grande	*Oryctolagus cuniculus*	2002	successful	T, P	29
Macaronesia	POR	Selvagem Grande	*Mus musculus*	2003	successful	P	17
Macaronesia	POR	Selvagem Grande	*Capra hircus*	1900	successful		4
Macaronesia	SPA	Alegranza	*Felis Catus*	2002	successful	T, P	14
Macaronesia	SPA	Gran Canaria	*Acridotheres tristis*	2006	successful	T	16
Macaronesia	SPA	Isla de los Lobos	*Felis Catus*	2002	successful	T, P	14
Macaronesia	SPA	Montana clara	*Oryctolagus cuniculus*	2001	successful	T	14
Macaronesia	SPA	Tenerife	*Acridotheres tristis*	2000	successful	S, T	16
Mediterranean Sea	FRA	18 islets	*Rattus rattus*	2000	successful	TP	22,30
Mediterranean Sea	FRA	Grand Congloué	*Rattus rattus*	1999	successful	TP	10,18
Mediterranean Sea	FRA	Grand Congloué	*Rattus rattus*	1995	unsuccessful	T, P	23,39
Mediterranean Sea	FRA	ilot de la Folaca	*Rattus rattus*	2001	successful	T, P	17
Mediterranean Sea	FRA	Ilot de la Folaccheda	*Rattus rattus*	2001	successful	T, P	17
Mediterranean Sea	FRA	Lavezzu	*Rattus rattus*	2000	successful	T, P	17
Mediterranean Sea	FRA	Lavezzu	*Capra hircus*	1994	successful	S, T	4
Mediterranean Sea	FRA	Petit Congloué	*Rattus rattus*	1999	unsuccessful	P	10
Mediterranean Sea	FRA	Petit Congloué	*Rattus rattus*	2005	successful	T, P	9,18
Mediterranean Sea	FRA	Plane	*Rattus rattus*	2005	successful	T, P	17
Mediterranean Sea	FRA	Toro	*Rattus rattus*	1991	successful	P	17
Mediterranean Sea	GRE	Atalanti	*Capra hircus*	1979	successful		4
Mediterranean Sea	GRE	Cyprus	*Rhynchophorus ferrugineus*	2009	being confirmed	T, P, H	47
Mediterranean Sea	GRE	Kasidis	*Rattus rattus*	2005	successful	P	17
Mediterranean Sea	GRE	Kastronisia-1	*Rattus norvegicus*	2006	successful	P	17
Mediterranean Sea	GRE	Kastronisia-1	*Rattus rattus*	2006	successful	P	17
Mediterranean Sea	GRE	Kastronisia-2	*Rattus norvegicus*	2006	successful	P	17
Mediterranean Sea	GRE	Kastronisia-2	*Rattus rattus*	2006	successful	P	17
Mediterranean Sea	GRE	Koufonisi (Lefki)	*Capra hircus*	1976	successful	S	4
Mediterranean Sea	GRE	Lachanou	*Rattus rattus*	2005	successful	P	17
Mediterranean Sea	GRE	Polemika	*Rattus rattus*	2005	unknown	P	17
Mediterranean Sea	ITA	Capraia	*Ailanthus altissima*	2001	uncompleted	HR, H	12
Mediterranean Sea	ITA	Gemino di Fuori (Elba)	*Rattus rattus*	2000	successful	P	17
Mediterranean Sea	ITA	Gemino di Terra (Elba)	*Rattus rattus*	1999	successful	P	17
Mediterranean Sea	ITA	Giannutri	*Rattus rattus*	2007	successful	P	5,38
Mediterranean Sea	ITA	Isola dei Topi	*Rattus rattus*	2000	reinvaded	P	17

Appendix 1 continued

Region	Country	Name of island	Invasive species	Erad year	Eradication status	Methods	Ref code
Mediterranean Sea	ITA	Isola delle femmine	*Rattus norvegicus*	2009	successful	P	41
Mediterranean Sea	ITA	Isola delle femmine	*Oryctolagus cuniculus*	2009	uncompleted	T	21,26,41
Mediterranean Sea	ITA	Isola delle femmine	*Opuntia ficus-indica*	2002	successful	HR	41
Mediterranean Sea	ITA	Isola delle femmine	*Solanum sodomaeum*	2006	successful	HR	41
Mediterranean Sea	ITA	Isola La Scola	*Rattus rattus*	2001	successful	P	17
Mediterranean Sea	ITA	Isolotto d'Ercole	*Rattus rattus*	2000	reinvaded	P	17,26
Mediterranean Sea	ITA	Molara	*Rattus rattus*	2008	being confirmed	P	8,26
Mediterranean Sea	ITA	Pianosa	*Felis catus*	2007	uncompleted	T	15
Mediterranean Sea	ITA	Procida	*Ceratitis capitata*	1970	unsuccessful		49
Mediterranean Sea	ITA	Scoglio La Peraiola	*Rattus rattus*	2000	reinvaded	P	17
Mediterranean Sea	ITA	Zannone	*Rattus rattus*	2007	successful	P	5,38
Mediterranean Sea	SPA	Conills (Ibiza)	*Rattus rattus*	1999	successful	P	17
Mediterranean Sea	SPA	Dragonera (Mallorca)	*Capra hircus*	1975	successful	S	4,14
Mediterranean Sea	SPA	Isla grossa	*Oryctolagus cuniculus*	1993	unknown		14
Mediterranean Sea	SPA	Mallorca	*Acridotheres tristis*	2007	successful	S, T	16,19
Mediterranean Sea	SPA	Menorca	*Oxyura jamaicensis*	2001	successful	S	19
Mediterranean Sea	SPA	Menorca	*Carpobrotus edulis*	2005	uncompleted	HR	43
Mediterranean Sea	SPA	Ray Francisco (Isla del Rey)	*Rattus rattus*	1992	successful	P	17
Mediterranean Sea	SPA	Ray Francisco (Isla del Rey)	*Rattus rattus*	2000	successful	P	17
N Atlantic Ocean	DEN	Anholt	*Pinus mugo*	2005	being confirmed	HR	42
N Atlantic Ocean	DEN	Læsø	*Pinus mugo*	2005	being confirmed	HR	42
N Atlantic Ocean	EST	Hiiumaa	*Neovison vison*	1999	successful	T	14
N Atlantic Ocean	FIN	Korppoo	*Neovison vison*	2001	successful	S, T	28
N Atlantic Ocean	FIN	Nauvo	*Neovison vison*	2001	successful	S, T	28
N Atlantic Ocean	FIN	Trunsö	*Neovison vison*		on going	S, T	2
N Atlantic Ocean	FIN	Utö	*Neovison vison*		on going	S, T	2
N Atlantic Ocean	FIN	Vänö	*Neovison vison*		on going	S, T	2
N Atlantic Ocean	FRA	6 islets	*Rattus norvegicus*	2000	unsuccessful	T, P	17
N Atlantic Ocean	FRA	Bono	*Rattus norvegicus*	1994	successful	T, P	17
N Atlantic Ocean	FRA	Bono	*Capra hircus*	1993	successful	T	4
N Atlantic Ocean	FRA	Cézembre	*Rattus rattus*	2004	unsuccessful	T, P	23
N Atlantic Ocean	FRA	Chatellier	*Rattus norvegicus*	1994	successful	T, P	17
N Atlantic Ocean	FRA	Dumet	*Vulpes vulpes*	2003	successful	T	23
N Atlantic Ocean	FRA	Enez ar C'hrizienn	*Rattus norvegicus*	1996	successful	T, P	17
N Atlantic Ocean	FRA	Île aux Chevaux	*Rattus norvegicus*	2002	successful	T, P	17
N Atlantic Ocean	FRA	Ile aux Moines	*Rattus norvegicus*	1994	successful	P	17
N Atlantic Ocean	FRA	Ile aux Moines	*Capra hircus*	1993	successful	T	4
N Atlantic Ocean	FRA	Île aux Rats	*Rattus norvegicus*	1994	successful	T, P	17
N Atlantic Ocean	FRA	Île des Morts	*Rattus norvegicus*	2005	unsuccessful	TP	11
N Atlantic Ocean	FRA	Île Plate	*Rattus norvegicus*	1994	successful	T, P	17
N Atlantic Ocean	FRA	Kemenez	*Mustela putorius*	2003	successful	T	22,32
N Atlantic Ocean	FRA	Le Loc'h	*Rattus norvegicus*	2003	unsuccessful	T, P	17
N Atlantic Ocean	FRA	Ledenez Kemenez	*Mustela putorius*	2003	successful	T	22,32
N Atlantic Ocean	FRA	Rimains	*Rattus norvegicus*	1994	successful	T, P	17
N Atlantic Ocean	FRA	Rocher de Cancale	*Rattus norvegicus*	1994	successful	T, P	17
N Atlantic Ocean	FRA	Rouzic	*Rattus norvegicus*	1951	successful	P	17
N Atlantic Ocean	FRA	St. Riom	*Rattus norvegicus*	2000	unsuccessful	T, P	17
N Atlantic Ocean	FRA	Tomé	*Rattus norvegicus*	2002	successful	T, P	17
N Atlantic Ocean	FRA	Trébéron	*Rattus norvegicus*	2005	unsuccessful	TP	11
N Atlantic Ocean	FRA	Trielen	*Rattus norvegicus*	1996	successful	T, P	17
N Atlantic Ocean	FRA	Trielen	*Capra hircus*	1998	successful	T	4
N Atlantic Ocean	ICE	Flatey Island	*Rattus norvegicus*	1971	successful	P	17
N Atlantic Ocean	ICE	Flatey Island	*Mus musculus*	1971	successful	P	17
N Atlantic Ocean	IRE	Horse	*Capra hircus*	1994	successful		4
N Atlantic Ocean	POR	Bugio	*Oryctolagus cuniculus*	2008	being confirmed	P	29
N Atlantic Ocean	POR	Bugio	*Mus musculus*	2008	being confirmed	P	29
N Atlantic Ocean	POR	Bugio	*Capra hircus*	2008	being confirmed	P	29
N Atlantic Ocean	UK	Alisa Craig	*Rattus norvegicus*	1991	successful	P	14
N Atlantic Ocean	UK	Canna	*Rattus norvegicus*	2006	successful	P	48
N Atlantic Ocean	UK	Cardigan	*Rattus norvegicus*	1980	successful	P	17
N Atlantic Ocean	UK	Handa	*Rattus norvegicus*	1997	successful	P	17
N Atlantic Ocean	UK	Holy	*Capra hircus*	1963	unsuccessful		4
N Atlantic Ocean	UK	Jersey	*Lymantria dispar*		unknown	T	46
N Atlantic Ocean	UK	Lewis and Harris	*Neovison vison*		on going	T	36
N Atlantic Ocean	UK	Lundy	*Rattus norvegicus*	2004	successful	P	17
N Atlantic Ocean	UK	Lundy	*Rattus rattus*	2004	successful	P	17
N Atlantic Ocean	UK	Puffin (Seiriol's Island)	*Rattus norvegicus*	1998	successful	P	17
N Atlantic Ocean	UK	Ramsey	*Rattus norvegicus*	2000	successful	P	17
N Atlantic Ocean	UK	Uists	*Neovison vison*	2006	being confirmed	T	36
Pacific Ocean	FRA	Clipperton	*Sus scrofa*	1958	successful	S	37
Pacific Ocean	FRA	G'i	*Rattus exulans*	1998	successful	P	17
Pacific Ocean	FRA	Laregnere	*Rattus exulans*	1998	successful	P	17
Pacific Ocean	FRA	Le Prédour, Grande Terre	*Rattus rattus*	2010	on going		23
Pacific Ocean	FRA	Le Prédour, Grande Terre	*Oryctolagus cuniculus*	2010	on going		23
Pacific Ocean	FRA	Le Prédour, Grande Terre	*Cervus timorensis russa*	2010	on going		23
Pacific Ocean	FRA	Makapu	*Rattus exulans*	2003	unknown	P	17
Pacific Ocean	FRA	Mato	*Rattus rattus*	1998	successful	P	17
Pacific Ocean	FRA	Mekiro	*Rattus exulans*	2003	unknown	P	17

Appendix 1 continued

Region	Country	Name of island	Invasive species	Erad year	Eradication status	Methods	Ref code
Pacific Ocean	FRA	Motu-o-ari	*Rattus exulans*	2003	unknown	P	17
Pacific Ocean	FRA	Ndo	*Rattus exulans*	1998	successful	P	17
Pacific Ocean	FRA	Nge	*Rattus exulans*	1998	successful	P	17
Pacific Ocean	FRA	Otoi iti	*Rattus exulans*	2007	successful	P	23
Pacific Ocean	FRA	Redika	*Rattus exulans*	1998	successful	P	17
Pacific Ocean	FRA	Signal	*Rattus exulans*	1998	unsuccessful	P	17
Pacific Ocean	FRA	Surprise	*Mus musculus*	2005	successful	P	17
Pacific Ocean	FRA	Surprise	*Rattus rattus*	2005	successful	P	17
Pacific Ocean	FRA	Taere ere	*Rattus exulans*	2005	successful	P	23
Pacific Ocean	FRA	Taere ere	*Mus musculus*	2005	successful	P	23
Pacific Ocean	FRA	Teanaone and Tepapuri	*Rattus exulans*	2003	unknown	P	17
Pacific Ocean	FRA	Teuaua/Ua-Uka	*Rattus rattus*	1987	unsuccessful	P	17
Pacific Ocean	FRA	Teuaua/Ua-Uka	*Rattus exulans*	1988	unsuccessful	P	17
Pacific Ocean	FRA	Teuaua/Ua-Uka	*Rattus exulans*	1995	unknown	P	13,45
Pacific Ocean	FRA	Tiarao	*Rattus exulans*	2008	unknown	P	23
Pacific Ocean	FRA	Tiarao	*Rattus rattus*	2008	unknown	P	23
Pacific Ocean	FRA	Uatermbi	*Rattus exulans*	1998	successful	P	17
Pacific Ocean	FRA	Uatio	*Rattus exulans*	1998	successful	P	17
Pacific Ocean	FRA	Uie	*Rattus exulans*	1998	successful	P	17
Pacific Ocean	FRA	Uo	*Rattus exulans*	1998	successful	P	17
Pacific Ocean	FRA	Vahanga, Tuamotu	*Rattus exulans*	2000	unsuccessful	P	17
Pacific Ocean	FRA	Vua	*Rattus exulans*	1998	successful	P	17
Pacific Ocean	UK	Ducie	*Rattus exulans*	1997	successful	P	17,25
Pacific Ocean	UK	Oeno	*Rattus exulans*	1997	successful	P	17,25
Pacific Ocean	UK	Pitcairn	*Rattus exulans*	1998	unsuccessful	P	17
Pacific Ocean	UK	Pitcairn	*Felis catus*	1997	successful	S, T, P	27
S Atlantic Ocean	UK	Amy Island	*Rattus norvegicus*	2009	being confirmed	P	35
S Atlantic Ocean	UK	Ascension	*Felis catus*	2004	successful	S, T, P	33
S Atlantic Ocean	UK	Ascension	*Schinus terebinthifolius*	2009	being confirmed	HR, H	3,44
S Atlantic Ocean	UK	Ascension	*Ficus elastica*	2009	being confirmed	HR, H	3,44
S Atlantic Ocean	UK	Ascension	*Capra hircus*	1945	successful	S	4
S Atlantic Ocean	UK	Beaver Island	*Dusicyon griseus*	1999	unsuccessful	S, T, P	35
S Atlantic Ocean	UK	Beaver Island	*Felis catus*	1983	successful	S, T	35
S Atlantic Ocean	UK	Bottom Tussac	*Rattus norvegicus*	2001	successful	P	17,25
S Atlantic Ocean	UK	Calf Island	*Rattus norvegicus*	2001	successful	P	17
S Atlantic Ocean	UK	Calf Islet	*Rattus norvegicus*	2001	successful	P	17
S Atlantic Ocean	UK	Channel Island west	*Rattus norvegicus*	2007	successful	P	35
S Atlantic Ocean	UK	Double	*Rattus norvegicus*	2001	successful	P	17,25
S Atlantic Ocean	UK	Gough	*Arrhenatherum elatius*	2006	successful	H	3
S Atlantic Ocean	UK	Gough	*Sagina procumbens*		on going	HR, H	3
S Atlantic Ocean	UK	Gough	*Senecio burchellii*	1980	successful	HR	3
S Atlantic Ocean	UK	Gough	*Conyza sumatrensis*	1980	successful	HR	3
S Atlantic Ocean	UK	Governor Island	*Rattus norvegicus*	2008	being confirmed	P	35
S Atlantic Ocean	UK	Grand Jason	*Capra hircus*		successful		4
S Atlantic Ocean	UK	Grass Island	*Rattus norvegicus*	2000	successful	P	35
S Atlantic Ocean	UK	Green Island	*Rattus norvegicus*	2007	successful	P	35
S Atlantic Ocean	UK	Harpoon	*Rattus norvegicus*	2001	successful	P	17
S Atlantic Ocean	UK	Horse	*Rattus norvegicus*	2001	successful	P	17
S Atlantic Ocean	UK	Inaccessible	*Sus scrofa*	1950	successful	S	34
S Atlantic Ocean	UK	Inaccessible	*Phormium tenax*		on going	HR, H	3,20
S Atlantic Ocean	UK	Inaccessible	*Capra hircus*	1872	successful	S	4
S Atlantic Ocean	UK	Letterbox Island	*Rattus norvegicus*	2007	being confirmed	P	35
S Atlantic Ocean	UK	Little Coffin Island	*Rattus norvegicus*	2007	successful	P	35
S Atlantic Ocean	UK	Little Coffin Islet	*Rattus norvegicus*	2007	successful	P	35
S Atlantic Ocean	UK	Outer	*Rattus norvegicus*	2001	successful	P	17
S Atlantic Ocean	UK	Rat Island	*Rattus norvegicus*	2001	successful	P	17
S Atlantic Ocean	UK	Sedge Island	*Dusicyon griseus*	1970	successful	S, T	35
S Atlantic Ocean	UK	Skull Bay Island	*Rattus norvegicus*	2007	successful	P	35
S Atlantic Ocean	UK	Sniper Island	*Rattus norvegicus*	2009	being confirmed	P	35
S Atlantic Ocean	UK	St.Elena	*Equus asinus*		uncompleted		3,20
S Atlantic Ocean	UK	St.Elena	*Capra hircus*	1970	unsuccessful		4
S Atlantic Ocean	UK	Stick in the Mud	*Rattus norvegicus*	2007	successful	P	35
S Atlantic Ocean	UK	Tea	*Dusicyon griseus*	2008	successful	S, T, P	3,35
S Atlantic Ocean	UK	Tea	*Rattus norvegicus*	2009	being confirmed	P	35
S Atlantic Ocean	UK	The Knobs	*Rattus norvegicus*	2009	being confirmed	P	35
S Atlantic Ocean	UK	Top Tussac	*Rattus norvegicus*	2001	successful	P	17,25
S Atlantic Ocean	UK	Tristan da Cunha	*Felis catus*	1970	successful	S	1
S Atlantic Ocean	UK	Tristan da Cunha	*Capra hircus*	1951	successful	S	4

List of References: (1) Angel and Cooper 2006; (2) Banks *et al.* 2008; (3) C. Stringer pers. comm.; (4) Campbell and Donlan 2005; (5) Capizzi *et al.* 2006; (6) Chapuis *et al.* 2004; (7) Chapuis *et al* 2001; (8) D. Capizzi pers. comm.; (9) CEEP 2007; (10) Dupuis and Du Châtenet 2006; (11) Dutouquet and Hamon 2005; (12) F. Giannini pers. comm.; (13) Faulquier *et al*., 2009 (14) Genovesi 2005; (15) Giannini and Baldinelli 2008; (16) S. Saavedra pers. comm.; (17) Howald *et al.* 2007; (18) Tranchant *et al.* 2008; (19) J. Mayol pers. comm.; (20) K. Varnham pers. comm.; (21) Lo Valvo pers. comm.; (22) Lorvelec and Pascal 2005; (23) M. Pascal pers. comm.; (24) Lorvelec *et al.* 2004; (25) Martins *et al.* 2006; (26) N. Baccetti pers. comm.; (27) Nogales *et al.* 2004; (28) Nordstrom *et al.* 2002; (29) P. Olivera pers. comm.; (30) Pascal *et al*. 2005; (31) Pascal, 1980; (32) Abdelkrim *et al.* 2005; (33) Ratcliffe *et al.* 2010; (34) Ryan P. 2007; (35) S. Poncet pers. comm.; (36) S. Roy pers. comm.; (37) Lorvelec and Pascal M. 2006; (38) Sposimo *et al.* 2008; (39) Tranchant *et al.* 2007; (41) V. Di Dio pers. comm.; (42) www.nobanis.org; (43) Fraga *et al.* 2006; (44) P.Lambdom pers. comm.; (45) Pascal *et al.* 2009; (46) Meadows 2009; (47) Melifronidou - Pantelidou 2009; (48) http://www.ntsSeabirds.org.uk/properties/canna/canna_progress.aspx; (49) Harris 1975

Methods Code: S = Shooting; T = Trapping; HR = Hand Removal; P=Poisoning H = Herbicides

Introduced rodents in the Galápagos: colonisation, removal and the future

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Abstract Introduced rodents (ship rats (*Rattus rattus*), Norway rats (*R. norvegicus*) and mice (*Mus musculus*)) have been present in the Galápagos Islands for at least 300 years. Their presence has resulted in adverse effects on native flora and fauna, including the likely extirpation of native rodents. Control of rodents has mainly been to protect native species like the dark-rumped petrel (*Pterodroma phaeophygi*) and to reduce effects on human infrastructure. Introduced rodent eradication attempts in Galápagos have been conducted since the 1980s, generally on small islands, and mainly using poison bait either hand-laid or in bait stations. Successful eradications have all been of ship rats in drier years, when reduced vegetation biomass apparently restricts rat populations through food limitation. Eradication attempts are being planned for larger islands using aerial poison applications with a view to scaling up to islands as large as 57,000 ha.

Keywords: *Rattus rattus*, *Rattus norvegicus*, *Mus musculus*, islands, brodifacoum, eradication

INTRODUCTION

Introduced rats (*Rattus* spp.) and house mice (*Mus musculus*) are considered responsible for a significant number of extinctions and ecosystem changes on islands worldwide (Towns *et al*. 2006). Over the past 30 years, increasing success in eradicating rats from islands has often been followed by spectacular responses by resident populations of native species and re-colonisation by species that had been extirpated (Bellingham *et al*. 2010). These responses have led to increased eradication attempts on archipelagos worldwide. Although the size of islands where rodent eradications are attempted is increasing, there have been failures (Howald *et al*. 2007). Reviews of the impacts of rodents on islands, and the outcome of eradication attempts, provides information that can justify and inform plans for rodent eradications elsewhere and are therefore useful for eradication practitioners worldwide. In the tropics, more information on eradications of invasive rodents on islands is required and should include information about improving efficiency to reduce cost and assessing risks to non-target species (Howald *et al*. 2007; Harper *et al*. 2011). The aim of this paper is to briefly review the impacts of introduced rodents in the tropical Galápagos Archipelago, outline the eradication attempts to date, and assess techniques and risks for the future.

INTRODUCED RODENTS IN GALÁPAGOS

Three of the four species of rodents commonly introduced to oceanic islands have reached the Galápagos Archipelago (total area: 777,000 ha): ship rats (*Rattus rattus*), Norway rats (*R. norvegicus*) and house mice. The invasion history, and threats posed by introduced rodents to native flora and fauna of the Galápagos, are summarised below.

Ship rat

Ship rats were first introduced to Galápagos by pirates and whalers between 1600 and the 1700s. A population established at James Bay, Santiago Island (Fig. 1), where buccaneers careened their vessels. The first recorded specimen was collected at Santiago by Darwin in 1835 (Waterhouse 1839). Two subsequent waves of introductions were apparently associated with human colonisation of other islands in the archipelago (Patton *et al.* 1975). The first wave began in about 1830, when ship rats became established on Floreana and Isabela islands. The second wave began during the Second World War, when the rats became established on Baltra and Santa Cruz islands (Clark 1978) and were followed by introductions to smaller islands with increased human activity. Ship rats now inhabit 35 islands, which comprise 90% of the land area of the Galápagos.

Most of the knowledge about the impacts of rodents in the Galápagos relates to ship rats but even then information is scarce. Ship rats caused up to 70% reproductive failure in the dark-rumped petrel (*Pterodroma phaeophygi*), whose colonies are restricted to the highlands of Santa Cruz, Floreana, Santiago and Isabela islands (Cruz and Cruz 1987a, 1987b). On Punta Pitt, San Cristóbal Island, ship rats preyed on eggs and chicks of wedge-rumped storm-petrels (*Oceanodroma tethys*) and Madeiran storm-petrels (*O. castro*) leading to a dramatic decline in both populations (Valle 1996). Nesting success of the critically endangered mangrove finch (*Geospiza scandens*) was significantly higher where ship rats are controlled (B. Fessel pers. comm.). On Pinzón Island recruitment of

Fig. 1 Location of the Galápagos Islands and islands mentioned in the text.

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the endemic giant tortoise (*Geochelone elephantopus ephippium*) consistently failed due to predation of eggs and young by ship rats (McFarland *et al*. 1974). There is also evidence that invasions by ship rats were responsible for the extinction of several species of the endemic rice rats *Nesoryzomys* spp. and *Oryzomys galapagoensis* (Clark 1984).

Norway rat

Norway rats, were first introduced to Santa Cruz and San Cristóbal islands in the 1980s, were recently discovered on Rábida Island and may be on Isabela Island (Key and Muñoz 1994). This species has been slow to spread through the Galápagos, possibly due to the widespread distribution of ship rats, which on forested islands can displace Norway rats (Russell and Clout 2004; Harper 2006). Norway rats are very common in urban areas and are trapped in the highlands where water is more freely available (Key *et al*. 1994). Their effect on birds in the Galápagos is unknown, but it is likely to be adverse, considering the effects of Norway rats on land birds and seabirds elsewhere (Towns *et al.* 2006; Jones *et al*. 2008). Norway rats occupy approximately 20% of the land area of the Galápagos.

House mouse

Mice were possibly introduced at the same time as ship rats in the 17th century (Key *et al*. 1994) and are now found on 12 islands. However, some populations of mice may have been overlooked during monitoring for the larger rodents, as mice are often cryptic in the presence of rats probably due to interference competition (Harper and Cabrera 2009). Little is known of the impacts of mice in the Galápagos. They are known to affect numbers and recruitment of the cactus (*Opuntia echios*) by digging around roots and affecting their stability during periods of high rainfall when cacti often become waterlogged. This adverse effect is then exacerbated by land iguanas (*Conolophus subcristatus*), which subsequently eat cladodes from the toppled cacti (Snell *et al*. 1994).

Mice have the potential to affect birds in the Galápagos in similar ways to those reported for seabirds in the Southern Ocean (Angel *et al*. 2009), but this possibility has yet to be examined. Mice do eat and contaminate crops and damage infrastructure, thus having an economic impact on human activity. For example, mice have reportedly damaged the wiring in electronic equipment at Baltra Airport. Mice are present on at least 90% of the land area of the Galápagos.

Rodent control and eradication

So far, the control of rodents in the Galápagos has focussed on rats for species protection and to reduce damage to infrastructure and the contamination of food supplies. Ship rats were first controlled for species protection using poison in bait stations on Cerro Pajas, Floreana Island, in 1983 to protect a population of dark-rumped petrels (Cruz and Cruz 1987a). This programme has since been extended to other petrel colonies in the highlands of Santa Cruz, Santiago and San Cristobal. Rat control is also carried out on the north coast of Baltra Island to prevent them from reinvading the adjacent Mosquera and Seymour Norte Islands from which the rats have been eradicated (Harper *et al*. 2011). Rats are also controlled on Baltra at the airport, the military base, and at the refuse tip. Local authorities carry out control in urban areas on inhabited islands.

Attempts to eradicate ship rats from islands in the Galápagos began in the early 1980s (Table 1). Until now, they have been focused on smaller islands, but with the eradication of ship rats on Seymour Norte (Harper *et al.* 2011) planning is underway to attempt larger islands.

An early ambitious attempt to eradicate rats on a large island using bait dumps almost succeeded on Pinzón Island (Table 1). During a very dry year over 45 days in November and December, a team of 47 people established bait dumps at 50m spacing across the entire island (Cayot *et al*. 1996). Each bait dump comprised 200gm of Racumin (Coumatetralyl) powder combined with rice in a paper bag, which equates to an application rate of 1 kg poison/ha. Brodifacoum (Klerat) blocks were also hand broadcast

Table 1 Attempted eradications of ship rats (*Rattus rattus*) on islands in the Galápagos Islands, Ecuador.

Island	Size (ha)	Nearest main island	Distance to main island (m)	Year of eradication attempt	Technique	Poison Bait type	Success	Year Confirmed
Venezia	13.3	Santa Cruz	30	Early 1980s	unknown	unknown	No	-
Pinzon	1815	Santa Cruz	10,399	1988	Hand-laid bait dumps/ broadcast 50 x 50m grid	Racumin Klerat	No	-
Marielas Sur	1.3	Isabela	848	June 1988	Bait stations 25m x 25m grid	Klerat	Yes	1999
Marielas Norte	0.24	Isabela	812	June 1988	Bait stations 25m x 25m grid	Klerat	Yes	2009
Pitt	0.4	San Cristobal	622	1989	Hand broadcast/ trapping	1080	Yes	1989
Bainbridge Islands (4)	#1: 11.4 #3: 18.3 #5: 4.1 #6: 4.5	Santiago	#1 1024 #3 630 #5 1167 #6 874	2000	unknown	unknown	unclear	-
Lobos	6.7	San Cristobal	162	2002	Bait stations 30m x 30m grid	Klerat	No	-
Mosquera	4.6	Baltra	406	Early 1980s	unknown	unknown	No	-
Mosquera	4.6	Baltra	406	2007	46 bait stations	Klerat, 1080	Yes	2009
Seymour Norte	184	Baltra	1464	2007	Hand broadcast 25m x 25m grid	Klerat	Yes	2009

between bait dumps. On coastal cliffs Klerat blocks were thrown onto cliff faces. Most bait take was on the coast and in the more humid highlands where the last rat sign was in loose rocks on the crater walls. Monitoring in January, February, April, May, July-August, October (two trips) and November 1989 detected no rat sign from February until the end of October when sign was found at a single bait station. Although poison bait was laid around that bait station, more comprehensive sampling in November found sign of rats at 10 stations in the central highlands and higher southern slopes. These areas were re-poisoned with Racumin and Klerat (Cayot *et al*. 1996). By January 1990, the beginning of the 'hot' season and associated increase in rainfall made bait distribution untenable and the project was abandoned. Observed short-term benefits of rat suppression for native wildlife included increases in the abundance of juvenile marine iguanas (*Amblyrhynchus cristatus*) (Cayot *et al*. 1994) and in populations of endemic Pinzón lava lizards (*Microlophus duncanensis*) and Galápagos doves (*Zenaida galapagoensis*). Successful giant tortoise nesting was also recorded. There was an apparent decrease in the population of Galápagos hawks (*Buteo galapagoensis*) and short-eared owls (*Asio flammeus*) (Muñoz 1990).

One of the first successful eradications was on Pitt Island, an islet off San Cristóbal after ship rats colonised around 1983 (Valle 1996). The eradication attempt was confirmed successful in 1989 (Table 1).

In 2000, attempts were made to eradicate ship rats from the Bainbridge islands where they had established on four of the eight islands (Table 1). By 2002, no rats were detected on two of the four islands attempted, but the success within the island group is still unclear and requires extensive sampling to confirm the outcome.

DISCUSSION

There have been 10 recorded ship rat eradication attempts in the Galápagos since the early 1980s and five (50%) have been successful. The result from one operation at the Bainbridge Islands is unclear but appears to have mixed success, with some islands with rats still extant and one or two islands where rats have been eradicated. Most of the islands attempted have been small (< 20ha), although the successful eradication on Seymour Norte and failed Pinzón operation are exceptions.

Ship rats have been heavily suppressed or eradicated in the Galápagos with low poison application rates and this may be related to climatic conditions. On Pinzón approximately 1 kg/ha of Racumin was applied with rice as a bait which equated to 7.5g coumatetralyl/ha. Although there is no information on the rates of Klerat bait broadcast between Racumin bait dumps it appears that the application rates were relatively low. On Seymour Norte less than 3 kg/ha of Klerat bait was applied (Harper *et al*. 2011) which was equivalent to 150g brodifacoum/ha. In temperate islands applications routinely apply bait at rates of 12 kg/ha or more (Empson and Miskelly 1999; McClelland 2002) which equates to 240g brodifacoum/ha. In the Galápagos, the 1988 Pinzón Island eradication attempt, successful 1988 Marielas Islands, and 1989 Pitt Island eradications were carried out in particularly dry years. For example, in 1988 and 1989 78.5mm and 82.5mm annual rainfall respectively were recorded at Puerto Ayora (M. Gardener pers. comm.) instead of a median rainfall of 277mm. In contrast, an eradication attempt on Lobos Island in 2002 failed during a relatively wet year (577mm).

In the Galápagos, population densities of rats during dry years in all vegetation types rarely exceed five rats/ha whereas in particularly wet years densities reach 19 rats/ha (Clark 1980; Harper and Cabrera 2010). Ship rat populations on the Galápagos show food limitation with a positive correlation between population density and vegetation biomass (Clark 1980). The generally arid conditions that prevail in the Galápagos during the dry season and in non El Niño years thus appear to restrict ship rat populations. Strong food limitation for ship rats in the dry season was suggested by the apparent palatability of wax-based Klerat to the low density ship rat population on Seymour Norte (Harper *et al*. 2011). Failed rat eradications on tropical islands elsewhere were often timed at the end of wet seasons when abundant food was available (Rodríguez *et al*. 2006).

The information presented here suggests that relatively low poison bait application rates may be suitable for eradication attempts in dry years. Poison operations should be timed for the last three months of the dry season and in particularly dry years if possible. Low application rates will reduce resources and time required, as well as risks to non-target species, and should be tested on smaller islands in the Galápagos with a view to scaling up to larger operations.

Grid spacing of bait stations or hand-laid baits does not appear to have had any appreciable affect on the success of eradications although the sample size is small. Grids of ≤ 25m on three islands have all resulted in successful operations (Table 1). The grid spacing for the Pinzón operation was 50 x 50 but Klerat was hand sown between the bait dumps, effectively reducing the grid size.

Future operations

In April 2007, international rat eradication experts met in the Galápagos and drafted a plan, Project Pinzón, to eradicate rats from several larger islands in the archipelago (Cayot 2007). The plan included improving eradication experience in the Galápagos by beginning with rat eradications on smaller islands, then with the information and experience gained, scaling up eradication attempts to islands as large as Santiago (57,728 ha).

Since that meeting, rats have been eradicated on Seymour Norte. An operational plan has been completed for the eradication of ship rats on Pinzón Island and Norway rats on Rábida Island (499 ha) using aerially distributed brodifacoum 25D bait (Bell Labs) in late 2010 or 2011 (Harper 2009). The 2010 El Niño event may postpone the operation if it results in substantial vegetative growth and an associated increase in rat abundance which would threaten the success of operation. Some smaller islands will be treated concurrently, including Roca Beagle Sur (8.7 ha); Roca Beagle Oeste (4.3 ha); Bartolomé (124 ha); Bainbridge Islands No.3, No. 5, No. 6; and Plaza Norte (8.8 ha). All of these islands have ship rats except for Plaza Norte, which has mice.

Keeping islands rodent-free

The success of the planned eradications will depend in part on substantially improved biosecurity measures. There are substantial numbers of small boat journeys between Galápagos islands for tourism, domestic fishing, and personal travel. All of these journeys pose risks for further introductions to islands and reintroductions of rodents to islands where they have been eradicated. The development and implementation of biosecurity measures that can capture every boat journey and detect rodents as small as mice is a challenge but will be essential if Galápagos Islands are to remain free of introduced rodents.

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The history of mammal eradications in Hawai`i and the United States associated islands of the Central Pacific

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Abstract Many eradications of mammal taxa have been accomplished on United States associated islands of the Central Pacific, beginning in 1910. Commonly eradicated species are rabbits (*Oryctolagus cuniculus*), rats (*Rattus* spp.), feral cats (*Felis catus*), and several feral ungulates from smaller islands and fenced natural areas on larger Hawaiian Islands. Vegetation and avifauna have demonstrated dramatic recovery as a direct result of eradications. Techniques of worldwide significance, including the Judas goat method, were refined during these actions. The land area from which ungulates have been eradicated on large Hawaiian Islands is now greater than the total land area of some smaller Hawaiian Islands. Large multi-tenure islands present the greatest challenge to eradication because of conflicting societal interests regarding introduced mammals, mainly sustained-yield hunting. The difficulty of preventing reinvasion poses a persistent threat after eradication, particularly for feral pigs (*Sus scrofa*) on multi-tenure islands. Larger areas and more challenging species are now under consideration for eradication. The recovery of endangered Hawaiian birds may depend on the creation of large predator-proof exclosures on some of the larger islands. Large scale eradications of small Indian mongooses (*Herpestes auropunctatus*) would be beneficial to ground-nesting birds such as nēnē (*Branta sandvicensis*), but this has been achieved only in small exclosures.

Keywords: Carnivores, rabbits, recovery, rodents, ungulates, fences

INTRODUCTION

The terrestrial biota of the Central Pacific is defined by its degree of isolation. For example, the Hawaiian Archipelago is 3200 km from any continental land mass (Ziegler 2002). After tens of millions of years of evolutionary isolation from all mammals except bats, islands of the Central Pacific were besieged by rodents, carnivores and herbivores (Ziegler 2002). The first mammals were introduced by early canoe voyagers of the Pacific more than 1000 years ago (Kirch 1982). The discovery of the Hawaiian Islands by Cook in 1778, like many other islands of the Pacific, brought introductions of hoofed animals for beasts of burden, milk, hides and meat as well as additional species of rodent and predators to control rodents.

Ecological degradation ensued and groups of endemic plants and animals suffered extinctions, including flightless birds (Olson and James 1982; Steadman 1995), and nine percent of all Hawaiian flora (Sakai *et al*. 2002). After a century of settlement by westerners, the concept of eradicating non-native species came about as a solution to agricultural, public health, or economic problems (Tomich 1986), and more recently, to solve ecological problems (Hess *et al*. 2009). Reversing the effects of alien mammals has proven to be difficult, but successes have resulted in the recovery of native biota (Hess *et al*. 2009).

This paper reviews the history of invasive mammal management on United States associated islands of the Central Pacific, particularly as it involves eradications and the effects of these actions on native biota. Questions we address are: has the scale of eradications increased? Are additional species being eradicated? Are new techniques being developed and employed? We aim to provide perspective on the Central Pacific islands both in space and time, and how current and future management of invasive mammals compares to the past.

RESTORATION THROUGH ERADICATIONS

All eradications are listed in Table 1 and locations are given in Fig.1.

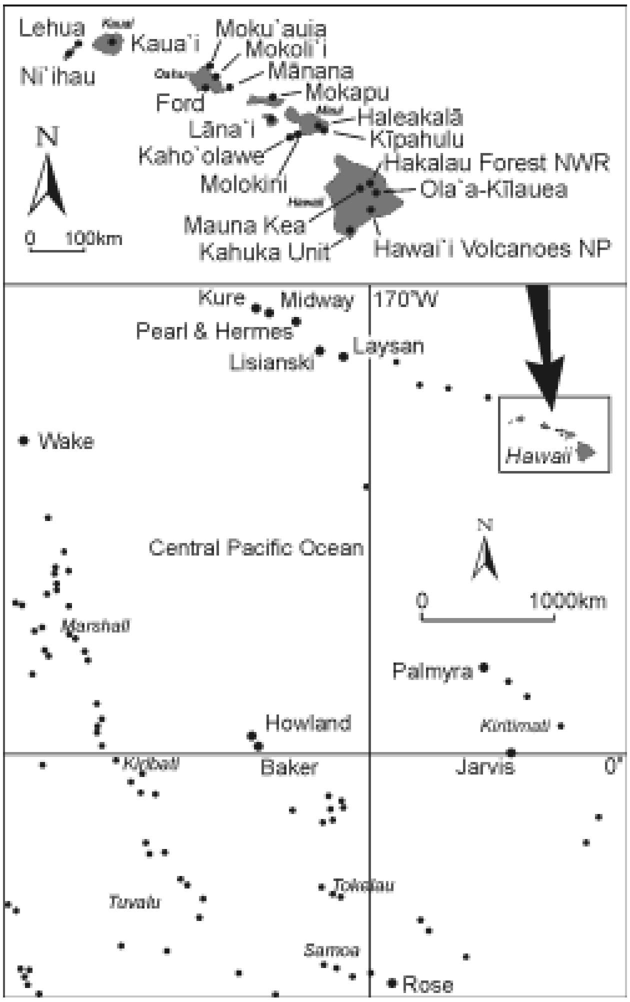

Fig. 1 Locations of mammal eradications from U.S. administered islands of the Central Pacific. Island group names (italicised) are included to provide location information.

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Table 1 Mammal eradications from U.S. administered islands of the Central Pacific.

Species Location	Area ha	Year Introduced	 Eradicated	Method
Rabbits				
Laysan	400	1902	1923	Shooting
Lisianski	170	1903	< 1923	Starvation
Pearl & Hermes Atoll	30	< 1916	1928	Shooting
Ford, Pearl Harbor	183	< 1825	?	Starvation?
Mānana, O`ahu	25	< 1890	< 1985	Starvation
Molokini, Maui	8	< 1915	< 1965	?
Haleakalā, Maui	25	1989	1990	Snaring, shooting, and live-trapping
Kaua`i	?	2000s	2003	Trapping
Lehua Islet, Ni`ihau	110	< 1930	2006	Dogs and hunters
Total	951			
Pacific rats				
Rose Atoll, Samoa	6.3	< 1920	1992	Brodifacoum bait stns, live- & snap-traps, bromethalin
Green Island, Kure	129	?	1993	Brodifacoum bait stations, live- & snap-traps
Moku`auia, O`ahu	385	?	2000	Diphacinone bait stations, live- & snap-traps
Mokapu, Moloka`i	7	?	2008	Diphacinone aerial broadcast
Lehua Islet, Ni`ihau	110	?	--	Diphacinone aerial broadcast in 2009
Total	584			
Ship rats				
Eastern Is, Midway	134	1940s	1994	Brodifacoum bait stations, live-traps, snap-traps
Spit Island, Midway	1	1940s	1994	Brodifacoum, Live-traps
Sand Island, Midway	486	1940s	1997	Brodifacoum bait stations, live-traps
Palmyra Atoll	275	1940s	--	Brodifacoum hand broadcast in 2001
Mokoli`i, O`ahu	5	?	2002	Diphacinone bait stations
Moku`auia, O`ahu	385	2004	2006	Diphacinone bait stations, live- & snap-traps
Total	1011			
Cats				
Baker	164	1937	1960s	Direct pursuit-hunting
Howland	184	1937	1986	Shooting, trapping
Jarvis	450	1885? 1937	Died out 1990	Shooting, trapping, poisoning, virus
Wake	737	1960s	2004	Shooting, trapping
Total	1535			
Pigs				
Lāna`i	36,130	> 1911	mid-1930s	Shooting
Kīpahulu Valley, Maui	1400	1970s	1988	Snaring
HAVO, Hawai`i	7800 (16,180)	1790s	1989 (2007)	Dogs, shooting, snaring
HFNWR, Hawai`i	4450	1790s	2004	Dogs, shooting, snaring
Ola`a-Kīlauea	14,120	1790s	1995-2010	Driving, trapping, shooting, snaring
Total	72,280			
Goats				
Ni`ihau	18,910	1900s	1910–11	Contract Hunting
Jarvis	450	?	1935	Self-extirpation
Lāna`i	36,130	1800s	1981	Ground shooting
HAVO, Hawai`i	55,440	1778	1984	Drives, shooting, Judas
Haleakalā NP, Maui	13,690	> 1780	1989	Drives, shooting, Judas
Kaho`olawe	11,650	1793	1990	Helicopter & ground shooting, Judas
Mauna Kea, Hawai`i	32,110	1778	--	Drives, helicopter shooting since 1934
Total	136,270			
Sheep				
Lāna`i	36,130	mid-1800s	1980s	Ground shooting
Kaho`olawe	11,650	1858	1990s	Helicopter & ground shooting, Judas
Mauna Kea, Hawai`i	32,110	1778	--	Drives, helicopter shooting since 1936
Kahuku Unit, Hawai`i	46,800	1968	--	Ground shooting, dogs, helicopter shooting since 2004
Total	47,780			
Cattle				
HFNWR, Hawai`i	44,050	1800s	2004	Dogs, shooting, snaring, helicopter shooting

Rabbits

In the Northwestern Hawaiian Islands, European rabbits (*Oryctolagus cuniculus*) were introduced as a source of food to Lisianski and Laysan islands about 1902, and subsequently discovered on Southeast Island of Pearl and Hermes Atoll in 1916 (King 1973).

Rabbits were eradicated from Laysan and Lisianski in 1923 after a failed eradication attempt on Lisianski in 1912–1913 (King 1973). Compounding the effects of mice (present since 1846), the rabbits eliminated most of Lisianski's vegetation by 1914, which then caused starvation of the rabbits (Olson and Ziegler 1995). Eradication of rabbits on Laysan coincided with desertification and the extinction of the Laysan honeycreeper (*Himatione sanguinea freethii*), the Laysan millerbird (*Acrocephalus familiaris familiaris*), and the last observations of Laysan rail (*Porzana palmeri*) (Ely and Clapp 1973). Rabbits were also eradicated on Southeast Island of Pearl and Hermes Atoll 1928 by shooting (King 1973; Amerson *et al.* 1974).

Among the larger Hawaiian Islands, rabbits were on Ford, Mānana, and Molokini islands, but disappeared, perhaps due to starvation (Swenson 1986). An incipient rabbit population was eradicated in Haleakalā National Park (HALE) on Maui in 1990 by shooting, trapping and snaring (Loope *et al*. 1992), and another on Kaua`i was eradicated by trapping in 2003 (C. Martin pers. comm.). Intensive hunting eradicated rabbits from Lehua Islet near the island of Ni`ihau in 2005–2006 (B. Keitt and C. Swenson pers. comm.). Rabbit releases have occurred on the larger Hawaiian Islands, without establishing wild populations.

Rodents

The Polynesian or Pacific rat (*Rattus exulans*) was among the earliest introductions of Pacific voyagers more than 1000 years ago (Kirch 1982; Matisoo-Smith and Robins 2004). House mouse (*Mus musculus*) reached the Hawaiian Islands by 1816 aboard European ships and Norway rats (*R. norvegicus*) were noted in Hawai`i as early as 1835, but ship rats (*R. rattus*) were not documented until 1899, apparently after the construction of shipping wharfs (Atkinson 1977). Introduced rodents, particularly ship rats, prey on birds at all life history stages and compete by preying on invertebrates and seeds, often interrupting reproduction in plants (Lindsey *et al*. 2009). The effects of Pacific rats may have included the disappearance of native lowland forests of Hawai`i in as little as 50 years (Athens 2009).

The first rat eradication in 1990, by the U.S. Fish & Wildlife Service (USFWS) and the Samoan Department of Wildlife and Marine Resources, was Pacific rats on 6.3 ha Rose Atoll, American Samoa. WeatherBlok containing 0.005% brodifacoum was used in bait stations spaced 50 m apart over the entire island, along with live- and snap-traps (Morrell *et al*. 1991; Ohashi and Oldenburg 1992). This eradication failed but a subsequent treatment with Vengeance (0.01% bromethalin, an acute neurotoxin) was successful (Murphy and Ohashi 1991).

In the Northwestern Hawaiian Islands, Wildlife Services (WS) of the U.S. Department of Agriculture's Animal and Plant Health Inspection Service and the Hawai`i Department of Land and Natural Resources (DLNR) eradicated Pacific rats in 1993 from 129 ha Green Island, Kure Atoll, using brodifacoum bait stations (J. Murphy pers. comm.). In 1994 the U.S. Navy, USFWS and WS eradicated ship rats from Eastern and Spit Islands at Midway Atoll (J. Murphy pers. comm.). Trapping and baiting with WeatherBlok of 134 ha Eastern Island was completed within three months. No evidence of rats was found at bait stations after a year (Murphy 1997a). The eradication of rats from 1 ha Spit Island in 1990 was accomplished within a month with live traps, incidental baiting and rat nest removal (J. Gilardi pers. comm.; Murphy 1997a).

The successful Eastern and Spit Island eradications, combined with evidence of the impacts rats were having on Bonin petrel (*Pterodroma hypoleuca*), persuaded the U.S. Navy to fund rat eradication on Sand Island (Seto and Conant 1996). In July 1996, the 486 ha island was overlaid with two 50 m grids, one for brodifacoum bait stations and one for live traps (Murphy 1997b). The last rat sighting was in October 1997. Sand Island remains the largest island and the only permanently inhabited island in the U.S. from which rats have been removed. Growth of the Bonin petrel population from an estimated 32,000 nesting birds (Seto and Conant 1996) to more than 900,000 provides compelling evidence for the enormous benefits of rat eradication. Native vegetation on Midway, such as naupaka (*Scaevola taccada*) and nohu (*Tribulus cistoides*), also became noticeably more dense and abundant (N. Hoffman pers. comm.). Mice on Sand Island are now the only small mammal remaining in the Northwestern Hawaiian Islands.

At Palmyra Atoll in the equatorial Line Islands, rats prevent six seabird species from nesting. An attempt to eradicate ship rats from the atoll by WS failed in 2001. This was the most complex eradication attempt by Hawai`i-based wildlife managers, involving approximately 275 ha and 54 islets, some of which were densely vegetated with coconut palms (*Cocos nucifera*), naupaka bushes and pāpala kēpau (*Pisonia grandis*) trees (Ohashi 2001). Numerous factors contributed to the failure, among them high rainfall in a complex forest habitat which resulted in rat foraging ranges that were smaller than the 50 m bait station spacing, and high bait take by land crabs *Cardisonma carnifex*, *Coenobita brevimanus* and *C. perlatus*. A successful pilot eradication on several small islets using hand broadcast of brodifacoum at a rate of 90 kg/ha was conducted in July 2005 after the failure was evaluated.

The successes of rat eradication on remote islands have also brought about efforts to restore offshore islets of the main Hawaiian Islands. In 2002, the Offshore Islet Restoration Committee was formed to restore selected islets around the Hawaiian Islands. To date, rat eradications have been successful on Moku`auia and tiny Mokoli`i Islet, both near O`ahu, using traps and diphacinone in bait stations (J. Eijzenga pers. comm.). Wedge-tailed shearwaters (*Puffinus pacificus*) subsequently began fledging from Mokoli`i, although ship rats have apparently reinvaded (D. Smith pers. comm.). A joint project by the USFWS, Hawaii DLNR and WS to eradicate Pacific rats from 7 ha Mokapu Island off Moloka`i in February 2008 was the first rat eradication using an aerial application of a registered rodenticide (diphacinone) for conservation purposes in the U.S. (P. Dunlevy pers. comm.). Attempting to build on this precedent, diphacinone pellets were also broadcast by helicopter for Pacific rats in January 2009 on 110 ha Lehua Islet, but the eradication was unsuccessful (VanderWerf *et al*. 2007; P. Dunlevy pers. comm.).

Carnivores

Domestic cats (*Felis catus*) arrived with the earliest European explorers (Tomich 1986). "Wild" cats had spread as far as the wilderness of Kīlauea by 1840 (Brackenridge 1841). Feral cats continue to present challenges to managers of natural areas on islands where they are known to prey on birds, but there is little prospect for island-wide eradication (Lindsey *et al*. 2009). Cat predation of nesting wedge-tailed shearwaters on O`ahu, has caused total loss of reproductive success (Smith *et al*. 2002).

Cats were eradicated from Baker Island in 1964, Howland Island in 1987, and Jarvis Island in 1990 (Rauzon *et al*. 2011). Hunting on Baker and Howland sufficed, but Jarvis also required trapping, poisoning, and feline panleucopaenia virus to a limited extent (Rauzon 1985). These eradications resulted in the recolonisation of five extirpated seabird species (Rauzon *et al*. 2002). Feral cat eradication was completed on Wake Atoll in 2004 by Marine Endeavors. Seabird diversity and abundance as well as Pacific rats increased in the absence of cats (Rauzon *et al*. 2008), and rat eradication by Island Conservation is planned.

The small Indian mongoose (*Herpestes auropunctatus*) was introduced to the Hawaiian Islands from Jamaica in 1883 and released to reduce rat populations in sugar cane fields on Hawai`i Island, O`ahu, Moloka`i, and Maui (Hays and Conant 2007). Mongooses may have been effective at reducing damage to sugarcane by Norway rats for a short period of time prior to the arrival of ship rats in Hawai`i (Atkinson 1977). Mongooses are now regarded only as pests and predators of ground-nesting birds, particularly nēnē (Hawaiian goose; *Branta sandvicensis*) and waterbird species (Stone and Loope 1987; Banko 1992). Without adequate prevention, mongooses may yet colonise Kaua`i and Lāna`i, the fourth and sixth largest Hawaiian Islands. Mongoose eradication has been achieved only in small exclosures.

Ungulates

Pigs (*Sus scrofa*) from Island Southeast Asia were the first ungulates introduced to Central Pacific islands by the earliest colonists more than 1000 years ago (Kirch 1982; Larson *et al*. 2005). The effects of pigs are widespread in Hawai`i, and throughout the Pacific region. In Hawai`i, pigs may have remained near commensal situations until the admixture of other strains brought by Europeans beginning in 1793 (Ziegler 2002).

Goats were established on Ni`ihau in the early 1900s and eradication by contract hunting became warranted by 1910 or 1911 (Kramer 1971). Lāna`i was also affected by excessive browsing and, by 1900, large areas were deforested by sheep and goats introduced in the mid-1800s (Hobdy 1993). Charles Gay began goat and sheep eradication on his Lāna`i ranch in 1902 and fenced the summit cloud forest to protect the watershed. The ornithologist George C. Munro came to run Gay's ranch in 1911 and spent much of his first decade there shooting sheep and goats. He also began eliminating pigs that had been released in 1911. Munro eradicated pigs from Lāna`i by the mid-1930s, feral goats by 1981, and feral sheep in the 1980s. Introductions of axis deer (*Axis axis*) in 1920, and European mouflon sheep (*Ovis gmelini musimon*) in 1954, continue to limit vegetation recovery on Lāna`i.

Feral sheep have repeatedly reached excessive densities on Mauna Kea, devastating the watershed and dry subalpine woodland environment. Foresters for the Territory of Hawai`i conducted sheep drives starting in 1934 that eliminated tens of thousands. The Mauna Kea Forest Reserve (MKFR) was fenced in 1935-1937 (Bryan 1937a) and nearly 47,000 sheep and over 2200 other ungulates were removed in the following 10 years by foresters and Civilian Conservation Corps workers using drives on foot and horseback (Bryan 1937b, 1947). Populations rebounded when sport hunting became a management goal of wildlife biologists after World War II and by 1960, the dire condition of the Mauna Kea forest was decried (Warner 1960). Despite this knowledge, European mouflon were hybridised with feral sheep and released between 1962 and 1966 to improve hunting opportunities (Giffin 1982). Scowcroft (1983), Scowcroft and Giffin (1983), and Scowcroft and Sakai (1983) used exclosures, aerial photography and studied tree size classes to demonstrate the effects of browsing and bark-stripping by sheep, cattle, and goats on the subalpine vegetation. U.S. Federal District court orders of 1979 and 1986 mandated the removal of goats and sheep to protect the endangered palila (*Loxioides bailleui*) that feed and raise their nestlings on māmane (*Sophora chrysophylla*) seed pods. More than 87,000 sheep have been removed from the MKFR over a 75-year period, but sheep are still far from being eradicated. The fence surrounding Mauna Kea has not been maintained and several hundred sheep are removed each year by aerial hunting from helicopters (Banko *et al*. 2009).

Goats had been removed from Hawai`i Volcanoes National Park (HAVO) on Hawai`i Island since 1927 but with no lasting effect due to reinvasion from the reservoir of animals in surrounding areas (Baker and Reeser 1972). Managers of Hawai`i's National Parks took action on the recommendation of the Leopold Report on Wildlife Management in National Parks (1963), which stated: "A visitor who climbs a mountain in Hawaii ought to see mamane trees and silverswords, not goats." The eradication of goats from 55,400 ha of the park took place from 1968 to 1984 (Tomich 1986). Goat eradication in HAVO proved the technical feasibility of eradicating ungulates from large areas of multi-tenure islands and developed specific techniques necessary to accomplish the task. The Judas goat method, which uses radio-telemetry to take advantage of gregarious behaviour in ungulates, has been replicated in many other management operations (Taylor and Katahira 1988). The re-invasion problem was solved by dividing areas into fenced units of manageable size, a difficult logistical process at the time for large areas and dense tropical forests on volcanic substrates. After a century and a half of degradation, a previously unknown endemic plant species, `āwikiwiki or *Canavalia kauensis* (now *C. hawaiiensis*), was found growing on the dry lowlands of Kukalau`ula in the absence of goats (St. John 1972).

At Haleakalā National Park (HALE) on Maui, 51 km of the 6920 ha Crater District was fenced between 1983 and 1987. Goats were also eliminated from the 4542 ha Kīpahulu District by the late 1980s (Stone and Holt 1990), and eradication of goats from the 13,700 ha park was completed in 1989 using techniques developed in HAVO (L. Loope pers. comm.).

Goats and sheep were eradicated from Kaho`olawe Island in 1990 by ground shooting, helicopter hunting, and the use of Judas animals (Kaho`olawe Island Conveyance Commission 1993). Goats and sheep had contributed to the loss of as much as 5 m of soil and interfered with livestock operations before the island became a bombing and shelling range after World War II (Kramer 1971).

The National Park Service was also the first to eradicate pigs from large areas of the Hawaiian Islands. Due to the steep terrain of Maui, feral pigs did not begin to invade the remote Kīpahulu Valley until the 1970s (Anderson and Stone 1993). Conventional control methods such as trapping and hunting dogs were precluded because helicopters were needed for access. Snaring was used to eradicate pigs from a 1400 ha area of Kīpahulu during a 45-month period beginning in 1978. Hunting dogs, shooting and snaring were also used to remove pigs from 7800 ha of HAVO from 1980–1989 (Katahira *et al*. 1993). The area from which pigs have been removed in HAVO increased to 16,200 ha by 2007 (D. Benitez pers. comm.). Native understory in the `Ōla`a Forest koa unit of HAVO increased 48% from 1991 to 1998, largely in the first two years following pig removal (Loh and Tunison 1999).

Hakalau Forest National Wildlife Refuge (HFNWR), also on Hawai`i Island, employed similar methods to remove pigs from a 4500 ha area in 1988–2004. Cattle were eradicated concurrently. The long period of time to complete removal was due in part to the large size of one management unit (> 2000 ha), interspersed areas of continued sustain-yield hunting, high densities of pigs, and relatively late use of snares (Hess *et al.* 2007). Preventing reinvasion into pig-free areas requires maintenance in perpetuity. Fences must be inspected monthly for damage and corrosive volcanic environments require fence replacement every 5–15 years.

The Nature Conservancy of Hawai`i (TNCH), the Natural Area Reserve System of the Hawai`i Division of Forestry and Wildlife, East Maui Watershed Partnership and the Three-Mountain Alliance of Hawai`i Island have all adopted and refined techniques for managing ungulates across larger landscapes. Many of these lands adjoin each other, thereby creating buffers or blocks of ungulate-free areas with high conservation value. While techniques to control and remove ungulates are well-established, some pose additional new threats. European mouflon have not yet reached their full distribution on Hawai`i Island and may invade conservation areas that have fences < 2 m tall. Axis deer populations are growing on Maui where they were introduced in 1960 (Tomich 1986). Game farms and ranches may inadvertently (and illegally) release additional ungulate species.

Perspective on Size of Eradications

We examined the area from which alien mammals have been eradicated to determine trends and consider whether eradications are increasing, decreasing, or unchanged over time. There has been no significant increase in the area from which rats (linear regression; coefficient = 0.018, F_{10} = 0.04, p = 0.851), rabbits (coefficient = -0.021, F_6 = 2.26, p = 0.193) and cats (coefficient = 0.150, F_3 = 6.62, p = 0.124) have been eradicated but cats show the strongest positive trend (r^2 = 0.77). The number of islands from which rabbits can be eradicated is now virtually zero. Rodent eradications have only recently begun in earnest. Despite the small number of islands from which cats have been eradicated, there appears to be an incipient pattern of application of successful techniques to larger islands. The trend in ungulates is more difficult to interpret because of incremental removal of contiguous populations on larger islands, repeated reinvasion, and lack of documentation (coefficient = -0.862, F_{13} = 0.36, p = 0.562). There were some unprecedented large-area ungulate eradications at a relatively early time, but later eradications have been of smaller areas.

Table 2 Areas from which ungulates have been eradicated in the Hawaiian Islands based on unpublished data from The Nature Conservancy of Hawai`i (TNCH). Other agency acronyms are: East Maui Watershed Partnership (EMWP), Hawai`i Division of Forestry and Wildlife (DOFAW), Kaho`olawe Island Reserve Commission (KIRC), National Park Service (NPS), National Tropical Botanical Garden (NTBG), Natural Area Reserve System (NARS), Three-Mountain Alliance (TMA), U.S. Fish & Wildlife Service (USFWS) and West Maui Mountains Watershed Partnership (WMMWP).

Island/Location	Agency	Area ha
Hawai`i		
HAVO NP	NPS	23,910
Hakalau NWR	USFWS	4240
Ōla`a-Kīlauea	TMA	35,030
Kona Hema	TNCH	3270
Pōhakuloa Training Area	U.S. Army	3000
Pu`u Maka`ala	NARS	1170
Kīpāhoehoe	NARS	580
Pu`u Wa`awa`a	DOFAW	100
Manukā	NARS	40
Pu`u O Umi	NARS	30
Ka`ūpūlehu	NTBG	30
Total		39,920
Maui Nui		
Kaho`olawe	KIRC	11,550
Haleakalā NP	NPS	10,610
West Maui	WMMWP	5760
East Maui	EMWP	2710
Waikamoi	TNCH	2180
East Maui	NARS	810
Auwahi	`Ulupalakua Ranch	10
Olokui, Moloka`i	NARS	680
Kūka`iwa`a Pen, Moloka`i	NPS	60
Total		34,340
Kaua`i		
Alaka`i	DOFAW	70
O`ahu		
Wai`anae Range	NARS	110
Wai`anae Range	U.S. Army	70
Pe`ahināi`a, Ko`olau Range	U.S. Army	50
Honouliuli	TNCH	70
Total		300
Grand Total		74,620

THE NEAR FUTURE FOR RECOVERY AND REINTRODUCTIONS

Eradications of rodents, cats, and rabbits from smaller islands of the Central Pacific have been beneficial to seabirds but there are a limited number of such islands. The restoration of landbirds and terrestrial biota depends on our ability to manage pests at the landscape level of larger islands. Societal values for hunting ungulates and harbouring outdoor pets necessitates expensive barriers to exclude these animals from pest-free refuges on multi-tenure islands. Careful planning and multiple pest management strategies may be used to maximise the area of pest-free refuges in relation to boundary perimeter that must be fenced. There is roughly 75,000 ha of ungulate-free area in the larger Hawaiian Islands (TNCH, unpubl. data; Table 2), but this comprises only about 19% of all forest bird habitat (Price *et al.* 2009). There is no significant area from which all mammalian pests have been eradicated. This presents obstacles to the reintroduction of native species which today exist only in captivity, such as the `alalā (Hawaiian crow; *Corvus hawaiiensis*) which requires large areas with diverse native understory food plants, and is susceptible to predation by rats and toxoplasmosis hosted by feral cats (Work *et al.* 2000). Successful reintroductions of species like `alalā back into the wild will depend on the ability of landowners and management agencies to establish and maintain large pest-free areas across ownership boundaries for the indefinite future.

CONCLUSION

The concept of eradication arose independently and at a relatively early time in the Central Pacific due to the necessity to protect fragile small-island ecosystems, forested watersheds and ranching operations on larger islands. Techniques of worldwide significance have been developed here, particularly during the eradication of ungulates. In their review of feral goat eradications on islands, Campbell and Donlan (2005) acknowledged the development of the Judas goat technique in Hawai`i (Taylor and Katahira 1988), but they made no mention of the goat-free areas created by this technique in the National Parks of Hawai`i, which are larger than the combined area of Ni`ihau, Lāna`i and Kaho`lawe. Although there is a negligible amount of area that is entirely pest-free on the larger Hawaiian Islands, many conservation agencies and landowners are developing methods and capacity for this goal and proposing larger island-wide eradications, such as cats and rodents from Kaho`olawe. There are now few remaining uninhabited small islands with alien mammals in the Central Pacific. Regulation of toxicants in the U.S. (Fagerstone *et al.*1990; Poché 1992) and conflicting societal interests between conservation, sustained-yield hunting and free-ranging pets continue to present challenges for the management of larger natural areas on multi-tenure islands. Future prospects for eradications over the entire area of the largest islands are limited, but there is potential for creating fenced areas free from ungulates on public lands, which are inhabited by much of the endemic Central Pacific biota.

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The Global Islands Invasive Vertebrate Eradication Database: A tool to improve and facilitate restoration of island ecosystems

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Abstract Islands are important for the conservation of biodiversity because they house 20% of terrestrial plant and vertebrate species, have suffered 64% of IUCN-listed extinctions and have 45% of IUCN-listed critically endangered species. Yet islands make up only about five percent of the earth's surface. The main cause of extinction and endangerment to biodiversity on islands is the presence of invasive vertebrates. Fortunately, many future extinctions can be prevented by eradicating invasive vertebrates from islands. To assess the current state of this conservation tool, we are compiling a global database of terrestrial vertebrate eradications from islands, including successes and failures. To date, in the Global Islands Invasives Vertebrate Eradication Database we have documented approximately 950 island eradication attempts involving 28 species of invasive vertebrates in 12 families. These are preliminary data and will be updated and checked for accuracy as part of the Island Invasives: Eradication and Management conference, Auckland 2010. Most eradication attempts have been of rodents (>350) and bovid ungulates (>160). Moderate numbers of eradication attempts have been of cats (>90), suid ungulates (>55), and rabbits (>45). Most projects have been on islands smaller than 500 ha (68%) and in temperate climates (72%). Targeting eradications on larger and more tropical islands would lead to the protection of more biodiversity. To this end, our vision is to maintain an accurate, web-accessible, regularly updated database that can be used to promote and improve the protection of island ecosystems by eradicating invasive vertebrates.

Keywords: Endangered species, threatened species, endemic species, biodiversity, alien species, extinction

INTRODUCTION

Islands are the epicentre of the extinction crisis. While islands make up only five percent of the earth's surface area, they support 20% of all biodiversity, including a disproportionately high level of endemic species (Kier *et al*. 2009). This biodiversity is particularly fragile and the vast majority of extinctions have been island species. For example, about 95% of bird, 90% of reptile and 70% of mammal extinctions have been on islands. These extinctions are primarily the result of the introduction of invasive vertebrates to islands. Fortunately, techniques to remove invasive vertebrates from islands are available and the practice is becoming an accepted conservation management tool. To better understand how this tool has been used, and to improve its future use, we developed, and are populating, a database of all vertebrate eradication efforts on islands (www.islandconservation.org/db).

The eradication of invasive vertebrates from islands is among the most challenging and beneficial actions land managers can take to restore islands and protect threatened species. Collating and understanding the lessons learned in previous efforts to eradicate invasive vertebrates are critical to improving and promoting this valuable conservation tool. Published global reviews of eradication efforts include regional approaches for all taxa (Clout and Russell 2006; Genovesi and Carnevali 2011; Lorvelec and Pascal 2005) and global approaches for individual taxa such as goats (*Capra hircus*; Campbell and Donlan 2005), cats (*Felis catus*; Nogales *et al*. 2004; Campbell *et al*. 2011), rodents (Howald *et al*. 2007), and mongoose (*Herpestes* spp.; Barun *et al*. 2011). These provide valuable reviews of the eradication efforts for these species and regions. Most importantly, these reviews provide land managers

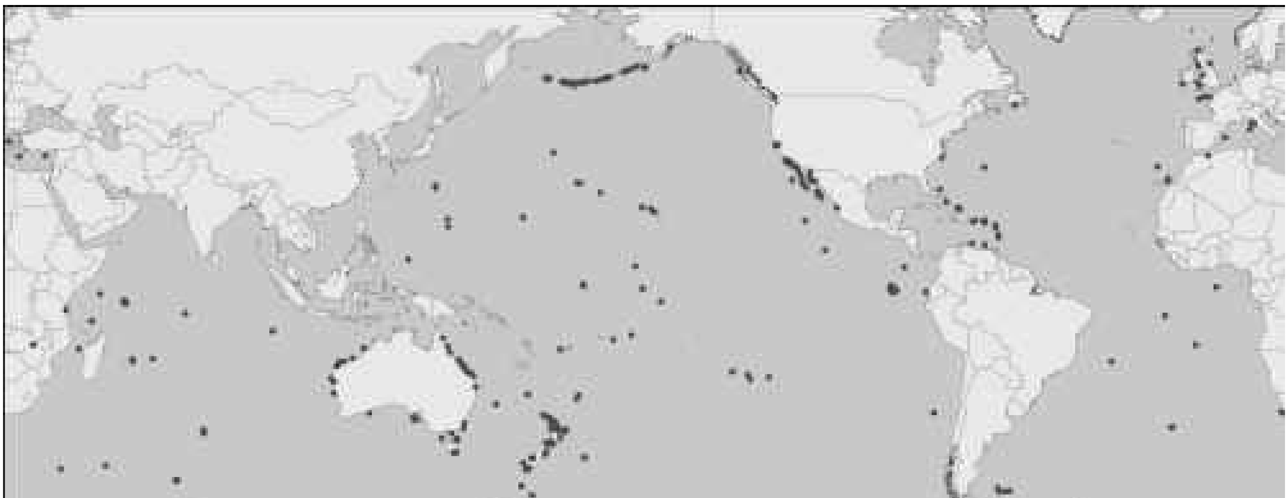

Fig. 1 Locations of all of the recorded eradications of invasive vertebrates from islands for which location data are available (n=664).

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Table 1 Invasive vertebrates in the database assigned to omnivore, carnivore and herbivore categories.

Omnivore	Carnivore	Herbivore
Gallirallus australis	*Alopex lagopus*	*Bos taurus*
Macaca mulatta	*Canis familiaris*	*Capra hircus*
Mus musculus	*Felis catus*	*Castor canadensis*
Rattus rattus	*Herpestes javanicus*	*Equus caballus*
Rattus exulans	*Mustela vison*	*Lepus nigricollis*
Rattus norvegicus	*Mustela erminea*	*Myocastor coypus*
Sus scrofa	*Mustela furo*	*Oryctolagus cuniculus*
Trichosurus vulpecula	*Mustela nivalis*	*Ovis aries*
	Procyon lotor	*Petrogale penicillata*
	Suncus murinus	
	Vulpes vulpes	

with information on which combinations of island size, technique, invasive species, and non-target species are feasible, and which combinations may have a high risk of failure. However, to date, there has been no global review of all vertebrate eradications on islands.

Here we present our vision for a web accessible database, including an initial analysis that provides details on eradication attempts including data on island characteristics, methods used, and contacts. Our goal is to highlight the most successful techniques, assess trends in eradication methods, and facilitate communication between practitioners to improve success. The database allows analysis of eradication effort for individual target species, and facilitates analysis of trends across different target invasive vertebrates.

It is important to note that this is an unfinished product, and we report here on preliminary data as of 15 December 2009. The Island Invasives: Eradication and Management Conference of February 2010 was used as a forum to validate and improve the database followed by a more thorough analysis and presentation at a later date.

METHODS

Data were mined from the published, grey, and unpublished literature. The bulk of the database came from the published summary articles for rodents (Howald *et al*. 2007), goats (Campbell and Donlan 2005), and cats (Nogales *et al*. 2004; Campbell *et al*. 2011). Additional data

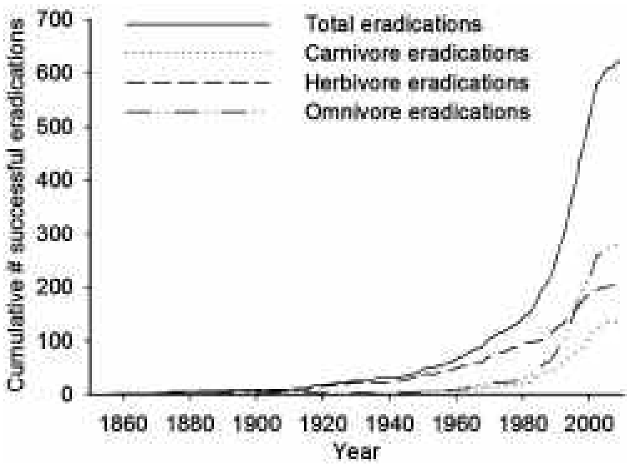

Fig. 2 Cumulative number of successful invasive vertebrate eradications on islands over time.

Table 2 Number of eradication attempts and success rate globally for select invasive vertebrates. An eradication event is defined as a successful or failed eradication attempt plus any follow up efforts on the same island.

Invasive vertebrate	Number of events	Failure rate %
Rattus	348	12.1
Goat	165	4.8
Cat	90	12.5
Pig	56	3.9
Rabbit	48	4.6
Fox	42	2.5
Mus	48	26.8
Mustelid	29	13.0*
Other	113	
Total	949	9.1

*50% of the eradication events in the database for mustelids list unknown for the eradication status so the reported failure rate is likely inaccurate for this group.

were collected through web searches, telephone interviews, emails, and specific requests directed at practitioners.

The database provides details of every documented eradication attempt, which is defined to include failures, successes, and follow up attempts on the same islands either after a failure or a reinvasion. Data categories were selected to provide information about each action, including specific details on methods, using drop down menus to facilitate analysis, and text fields to allow detail to be captured. For some analyses, all target invasive vertebrates were assigned a category of herbivore, carnivore, or omnivore (Table 1). Contact information and citations were provided where possible.

The methods used to populate the database have likely led to an underestimate of historical eradications, as those are less likely to be included in published papers or reports, and the people familiar with those projects are no longer involved in the field. The data also likely underestimate the failure rate for eradications, as failures are less likely to be reported. For these reasons, we tried to reach as many individual people as possible to encourage them to report older eradication efforts and failed eradications in the database.

Data on location (latitude and longitude), island size, country, and oceanographic region were extracted from the Global Islands Database (GID) (Depraetere 2007). For islands that were not in the GID we used the Meridian Data Global Island Database. Locations were verified using Google Earth and corrected if necessary.

RESULTS

As of 15 December 2009, we documented 949 vertebrate eradication attempts on islands globally (Fig. 1), involving 27 species of mammal and one species of bird. The three earliest documented eradication attempts were in 1673, 1686, and 1709. All three were of large ungulates and all three failed. The first documented successful eradication was of goats in 1857 on Norfolk Island, Australia.

Seven hundred and eighty six successful eradications were reported and 41 of those were later reinvaded. Fifty two eradications are listed as unknown, i.e. there was information indicating an eradication event took place but no data were available on the outcome, and eight were listed as incomplete. Ninety eradications were listed as failed

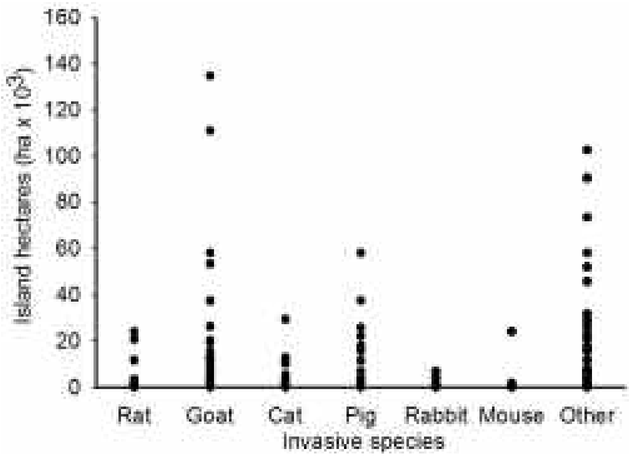

Fig. 3 Scatterplot showing area of islands where eradications have occurred for select species of invasive vertebrates.

eradication attempts. The success rate for all eradications with a known outcome was 91% (n=835, Table 2). Location data were available for 664 islands and the subsequent analyses that involve location data are restricted to these islands.

Since that first successful eradication over 150 years ago, rats (*Rattus* spp.) have become the invasive vertebrates most frequently eradicated from islands, with 348 reported eradication attempts, followed by goats with 165 eradication attempts (Table 2). The pace and scale of eradications have increased dramatically during this time (Fig. 2). After the first successful eradication in 1857 there were only 27 eradication attempts during the next 80 years (through 1940). From 1940-1980 there were 118 vertebrate eradication attempts, or about three per year. Since 1980, the rate of vertebrate eradications on islands has increased, with about 600 eradications between 1980 and 2009, or about 20 eradications per year (Fig. 2).

Along with increased frequency of eradications also came an increase in the size of islands from which invasive vertebrates were eradicated. The invasive vertebrate species that have been eradicated from the largest islands are goats, pigs and Arctic foxes (*Alopex lagopus*) (Fig. 3). Most of the largest islands had eradications implemented in the last 20 years (Fig. 4). Some of the attempts on large islands are near completion (e.g., removal of goats from Isabela, 412,000 ha). Other more ambitious island projects are being planned such as the eradication of rodents, cats and brushtail possums (*Trichosurus vulpecula*) from Stewart (170,000 ha) (Beaven 2008).

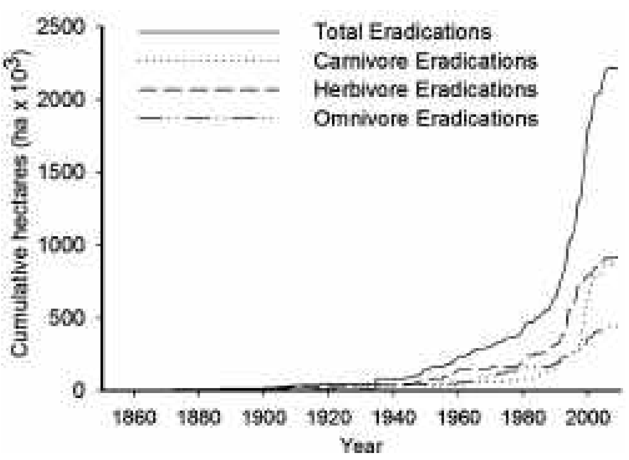

Fig. 4 Cumulative area in hectares of invasive vertebrate eradications over time for carnivore, herbivore and omnivore vertebrate eradications on islands.

Eradications have been attempted in 33 different countries, with New Zealand having 313 eradication events, followed by Australia with 154, and the United States with 139. France and Mexico have had 67 and 38 eradication events, respectively. The distribution of eradications is primarily in temperate regions. Of the 664 eradication events reported with latitudes for the islands, 436 have been attempted in temperate regions (23.5 to 60 degrees North and South latitudes) and only 180 in the tropics (between 23.5 and -23.5 degrees latitude). No eradications above 60 degrees latitude North or South were reported. Failure rate in the temperate regions was 7.6% (31 of 405) and 13.2% (21 of 159) in the tropics.

DISCUSSION

The first documented attempts to eradicate invasive vertebrates from islands were over 250 years ago, with the first successful attempt over 150 years ago in Australia. These early attempts to eradicate invasive vertebrate species began what is now a leading component of the conservation of island ecosystems and the protection of threatened species. Collecting details about current and historical vertebrate eradication attempts, including success rates, methods, costs, and island characteristics is required if this management tool is to be promoted and improved. The Global Islands Invasive Vertebrate Eradication Database project was designed to summarise information on all invasive vertebrate eradications and enable analyses that can: 1) help land managers and funders understand the applicability and limitations of eradication as a tool; 2) enable eradication practitioners to share information that facilitates iterative improvement, and 3) identify regions and target species for which eradication is under-utilised.

Preliminary analysis of the Global Islands Invasive Vertebrate Eradication Database indicates that the frequency of vertebrate eradications on islands is increasing. This demonstrates that conservationists, land managers, and funders have recognised and embraced the technique (Figs 2 and 4). Furthermore, the size of islands that have been attempted has increased. While not a perfect measure of cost, size of the island is positively linked to the cost of an eradication, thus the increase in size of islands with eradication is an indicator of the increased financial support for invasive vertebrate eradications from governments and funders.

New Zealand leads to protect island ecosystems, with 313 invasive vertebrate eradications attempted, which is more than the next three countries combined. This in part explains why a disproportionate number of eradications have been reported from temperate regions (Fig. 1). However, this concentration of eradications in temperate areas is unlikely to be the most efficient distribution of eradication effort to protect global biodiversity since most biodiversity is located in the tropics (Dirzo and Raven 2003).

Greater efficacy is also desirable in tropical latitudes. The rate of failed eradication efforts in the tropics is almost twice the rate in temperate areas. The reasons for this disparity are not known. However, the lack of seasonality in tropical environments may be a key factor. Many eradication campaigns take advantage of seasonal periods of reproduction and/or food stress for the target animal. For example, the over 40 Arctic fox eradications in the

Aleutian Islands, United States were undertaken during the winter when the target animal was primarily restricted to the coastlines (Ebbert 2000) The recommended strategy for rodent eradications is to apply bait when the target population is experiencing a food related, seasonal population decline (Howald *et al* 2007) and when reproduction is at its lowest. In tropical systems, these seasonal advantages are often more nuanced or completely absent.

It is not surprising that some invasive vertebrate species are harder to eradicate than others, based on success rates of eradication attempts (Table 2). Rodent eradications as a group experienced the highest failure rates, with 12.8%. This is likely due to the complexity of rodent eradications and the difficulty associated with putting every individual animal at risk during an eradication campaign. Surprisingly, at 12.5%, cat eradications had a similar failure rate to rodents. This is likely due to both the difficulty of detecting small numbers of cats on an island and the ability of cats to learn avoidance of available eradication techniques. The high failure rate for cats suggests a tendency among practitioners to underestimate the effort necessary to complete an eradication.

Invasive vertebrate eradication is becoming an increasingly accepted pathway to restoring native species and ecosystems, and is increasing in frequency, geographic distribution, size, and complexity. The Global Islands Invasive Vertebrate Eradications Database is designed to provide context for what types of eradications are simple or challenging and also to encourage communication between experienced practitioners and land managers that are protecting biodiversity on islands. It should not only be used by eradication practitioners, but also by island land managers, government agencies and foundations. However, its ongoing utility depends on everyone who conducts an eradication taking the time to input their own work and review other relevant entries.

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What does it take to eradicate a feral pig population?

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Abstract Control of feral pigs (*Sus scrofa*) has become a high priority for management of many island and mainland ecosystems, but few programmes have used population models to estimate the effect of harvest intensity on population size. We used data collected from 1991 to 1999 on density and sex/age structure of feral pigs on Santa Cruz Island, California, to develop a Leslie matrix model for estimation of the likelihood of eradication and number of years to eradication for different combinations of harvest rates and initial population size (N_0). The model included an estimated island-wide carrying capacity (*K*) of 3400, annual harvest rates of 0-95% for all sex and age classes, a management programme duration of ten years, and three levels of N_0: 25% of *K* (low population), 75% (average population), and 150% (high population). The rate of reduction in population size depended on N_0 at low to moderate harvest rates (5%-65%) but not high harvest rates (>70%). Mortality from harvest shifted from compensatory to additive once harvest rates > 10%, but population size tended to stabilise, albeit at substantially reduced levels, for annual harvest rates < 70%. Harvest rates between 60% and 70% reduced the population to low enough numbers that pigs could be considered ecologically extinct, but there was no likelihood of eradication until 70-75% of the population per year was harvested. Once this threshold was crossed, the likelihood of eradication increased rapidly to 1 for all N_0's. The median number of years to eradication when harvest rates > 70% ranged from ten (72% annual harvest rate) to 2.5 (95% harvest rate). The simulations suggest that N_0 will not add appreciable amounts of time to eradication programmes when harvest rates are high and that a strategy of intense harvest for five years will likely achieve eradication of many insular feral pig populations.

Keywords: Conservation, demography, invasive species, islands, feral animals, Leslie matrix, population management, population models, *Sus scrofa*

INTRODUCTION

The effects of non-native vertebrates on insular ecosystems have been recognised for decades (Atkinson 1989; Simberloff 1995; Mack e*t al.* 2000). These include altered ecosystem processes (Fukami *et al.* 2006), destruction or degradation of vegetation communities (Coblentz 1978), altered trophic interactions (Fritts and Rodda 1998), and extinctions (Sax and Gaines 2008). Consequently, control or eradication of introduced species is widely regarded as being an integral step in conservation of island ecosystems (Myers *et al.* 2000; Veitch and Clout 2002; Courchamp *et al.* 2003).

Pigs (*Sus scrofa*) have been among the most devastating species introduced to island and mainland systems (IUCN 2005). They can cause long-term damage to crops (Geisser and Reyer 2004) and have been implicated in alterations to ecosystem, community, and species-level properties (Aplet *et al.* 1991; Cushman *et al*. 2004). Because of their impacts on natural and agricultural systems, control of pig populations has become increasingly common in many parts of the world (Choquenot *et al.* 1996; Bieber and Ruf 2005) and there has been an upsurge in efforts to eradicate them where possible, especially on islands (Lombardo and Faulkner 2000; Kessler 2002; Cruz *et al.* 2005).

Increased control and eradication efforts have resulted in sophisticated methods for programme planning, design, implementation, and monitoring (Morrison *et al.* 2007; Nogueira *et al.* 2007). Particular emphasis has been on methods for deciding when eradication has been achieved (Ramsey *et al.* 2009; Rout *et al.* 2009). Surprisingly, there has been less attention paid to the question of what harvest rates are necessary to achieve control or eradication. Determining what level of harvest can be economically sustained for a given period of time is crucial for determining if there are adequate resources for eradication, long-term control, or neither. First principles of population growth suggest that increasingly higher rates of harvest will likely lead to lower levels of abundance, greater probability of control or eradication, and shorter programme duration. But these harvest rates are unknown, as is the approximate point where mortality from hunting ceases to be compensatory and becomes additive, how initial population size influences the likelihood of control or eradication, what levels of abundance can be expected to result from a given harvest rate, and how long an eradication programme will take to complete.

Feral pigs have had especially acute effects on California's Channel Islands, where they were introduced to the four largest islands in the 19th century (Knowlton *et al*. 2007). On Santa Cruz Island (SCI), the largest of the eight California islands, pigs were first recorded in 1852 (Schuyler 1988). Their long term effects have been increased erosion rates, alteration of native vegetation communities, damage or destruction of endemic plant populations, reduced abundance of some vertebrate species, and impacts to archaeological sites (NPS 2003). Pigs co-existed with feral sheep on SCI for at least 150 years, but there is little evidence of negative interactions between the two species (Klinger 2007). Historical accounts (Daily 1989, 1994), qualitative surveys conducted before sheep were eradicated in the 1980s (Baber 1982; Van Vuren 1994), and observations of island residents all indicate that the pigs were at times very abundant.

The Nature Conservancy (TNC) conducted a trial eradication of pigs in a fenced portion (2250 ha) of SCI from 1989 to 1991 to evaluate the feasibility of eradication throughout the island (Sterner and Barrett 1991). Despite the success of this trial (Sterner and Barrett 1991), TNC decided not to proceed with wide scale eradication at that time (Klinger 2007). Instead, data would be collected in a systematic monitoring programme to improve estimates of pig abundance (Sterner and Barrett 1991) and to model their population dynamics.

In this paper, we used a nine-year dataset from the SCI monitoring to develop a matrix population model of the influence of varying harvest rates on abundance of feral pigs for three different initial population sizes. Matrix

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models are a common and powerful tool for analysing the relationship between the dynamics and vital rates of a population (Leslie 1945, 1946; Caswell 2001). To date, they have only been applied in a limited capacity to gain insight into population dynamics of pig populations (Neet 1995; Bieber and Ruf 2005), and none have been explicitly developed in the context of an eradication programme. Our goals were to use predictions from the models to help answer questions a manager might ask when designing a pig management programme: 1) what level of annual harvest is required to achieve eradication; 2) how long will it take to achieve eradication; and, 3) what is the effect of initial population size on the likelihood of achieving eradication?

MATERIALS AND METHODS

Santa Cruz Island

Santa Cruz Island (249 km^2) is 40 km off the southern California coast. Although the highest point on the island is only 741 m, topography is very rugged. Two east-west trending mountain ranges flank a long central valley, with the interior and exterior flanks of each range cut by numerous small, deep drainages.

Climate on SCI is Mediterranean with warm, dry summers and cool, wet winters. Summer temperatures typically range from 27° to 35° C, while winter temperatures generally range from 5° to 15° C. Approximately 80% of the precipitation falls from November through April (L. Laughrin, UC Natural Reserve System, unpublished data). Inter-annual variation in precipitation is relatively high; the mean annual rainfall from 1903-1999 was 50.5 cm with SD ± 23.4. The complex topography and soils on SCI have resulted in a diverse array of vegetation communities that are structurally similar to communities on the mainland (Brumbaugh 1980; Minnich 1980; Junak *et al*. 1995). The dominant vegetation communities include grasslands, chaparral, coastal scrub, woodland, and bishop pine (*Pinus muricata*) forest (Junak *et al*. 1995).

American Indians were the first human inhabitants on SCI, beginning approximately 9000 YBP (Glassow 1980). From the early 19th through latter 20th century SCI passed through a series of Spanish and American owners. The predominant land uses during this period were ranching and agriculture. Since the late 1970s, the island has been managed primarily as a conservation site by TNC and the National Park Service (NPS).

Human infrastructure on SCI is very limited; there are several small facilities in the central valley and the east and west ends of the island. A series of unpaved roads and trails provides access to 75% of the island; most of the northwestern 25% of SCI has no maintained roads or trails.

Pig abundance surveys and density estimation

Density estimates of feral pigs were derived from surveys conducted along 15 transects established on the western 90% of the island. The surveys were conducted during the wet season (late November through early March) each year from 1990 through 2000. The steep and irregular topography would have made cross-country transects impractical, therefore nine transects were established along existing roads and the other six on trails or abandoned roads. The 15 transects were selected randomly from a pool of 56 potential routes and varied in length from 2.9 to 20.4 km (Table 1). The order in which the surveys were conducted was randomised each year, including when repeat counts were conducted on the same transects in the same year. The surveys were conducted by a single observer on foot or in a vehicle. Observers on foot walked at a pace of 3-5 km/h; on surveys done from vehicles a single person would observe while another person drove the vehicle at a rate of 10-20 km/h. The data collected on the counts included the transect bearing, the distance and bearing to each group of pigs, the number in each group, and vegetation types where the groups were seen (Buckland *et al*. 2001).

The sighting distance and angle were used to derive the perpendicular distance of groups to the transect (Buckland

Table 1 The number of surveys per transect collected in each of ten seasons for estimation of feral pig density on Santa Cruz Island, California, 1990 – 2000. The counts were conducted from late November – early March each year (Season). Total is the number of transects surveyed (including repeat counts on the same transect), Length is the number of kilometres surveyed (including repeat counts on the same transect), Observed is the total number of pig groups sighted that season, and the Encounter rate is the mean number of groups observed per km (± SE).

Transect	Length (km)	Season									
		Wet 90/91	Wet 91/92	Wet 92/93	Wet 93/94	Wet 94/95	Wet 95/96	Wet 96/97	Wet 97/98	Wet 98/99	Wet 99/00
1	20.4	2	2	2	1		2	1		1	
2	15.0	2	2	2	2		1	2	1	1	1
3	14.7	2	2	2	2	1	1	2	1	1	1
4	18.4	2	2	2	2	1	2	1		1	
5	7.6	2	2	2	2	1	1	1	1	1	1
6	13.0	2	2	2	1		1	1			
7	9.3	2	2	2	2		1	1	1		
8	9.8	2	2	2	2		2	1	1	1	
9	12.1	2	3	3	2		1	1		1	
10	5.3	2	3	3	2		1	1		1	
11	3.4	2	3	3	1	1	2	1			1
12	2.6	2	3	2	2		1	1	1		1
13	4.3	2	2	2	2		1	2	1	1	
14	2.9	2	2	3	2	1	1	1	1		
15	6.1	2	2	3	2	1	1	1	1		1
Total		30	34	35	27	6	19	18	9	9	6
Length (total)		289.8	313.2	319.6	253.0	53.1	196.9	178.9	72.3	107.6	49.4
Observed		71	114	85	91	106	89	81	57	118	88
Encounter rate		0.24 ± 0.01	0.36 ± 0.01	0.27 ± 0.01	0.36 ± 0.02	1.99 ± 0.24	0.45 ± 0.03	0.45 ± 0.02	0.79 ± 0.05	1.1 ± 0.09	1.66 ± 0.12

et al. 2001). The distribution of the perpendicular distances were then used to model density with the programme DISTANCE (Buckland *et al.* 2001). Two key functions (uniform and half-normal) with cosine and polynomial expansion terms were used to generate and compare different models of density. We produced an initial set of models based on ungrouped perpendicular distances. If the fit of these models was inadequate (based on visual inspection of the observed and estimated distributions), we then grouped the data into intervals to improve model fit. The model with the lowest value for the corrected Akaike Information Criteria (AIC_c) was considered the one with the most support. When ΔAIC_c was < 2 then model selection was based on the visual fit of the model as well as χ^2 values for model fit.

Pig sex and age data

Data on pig population structure were collected during systematic hunts augmented with opportunistic kills. Hunting was conducted an average of 7-10 days per year in each of nine geographic zones in the western 90% of the island (Table 2). The hunts were conducted in all months of the year. From 1990 through 1994, all hunts were conducted with 1-6 Catahoula Leopard Stock Dogs working with hunter groups. From 1995-1998 a single Catahoula was used on the hunts. Hunter groups were comprised of trained volunteers, NPS staff, and members of two municipal southern California Special Weapons and Tactics (SWAT) teams. Hunter/dog teams would sweep individual drainages within a watershed and kill all pigs flushed out, regardless of size or coloration. Field necropsy was done on all kills to determine sex, age class (years), body condition (indexed by the thickness of rump fat), and reproductive status. Data collected for reproductively active females included the number of foetuses, the number of lactating teats, or the number of piglets accompanying her. Age was determined by patterns of tooth wear and eruption (Matschke 1967).

Population modelling

A two-step process was used to model the effect of different harvest rates on the pig population. First, a base model was developed to determine if parameter estimates derived from the kill and density data were biologically realistic. We knew from historic records that pigs had persisted on SCI for at least 150 years, but had pronounced fluctuations in density over this period. We reasoned that a realistic model of the population would be highly variable over a 150 year period, but there would be no extinction events or abnormally high densities. Once we developed a biologically realistic base model, our second step would be the addition of annual harvest rates over a period of time representative of most pig eradication programmes.

Table 2 Effort and success rates for feral pig hunts on Santa Cruz Island, California. Days is the total number of days each year when hunts were conducted, Hunters is the mean number of hunters per hunt, Success is the number of hunts where at least one pig was killed, and Kills is the total number of pigs killed where data on sex and age class were collected [1].

Year	Days	Hunters	Hunter-Days	Success	Success (%)	Kills
1990	9	1	9	8	88.9	16
1991	65	2.8	182.0	56	86.2	109
1992	73	2.4	175.2	71	97.3	276
1993	68	2.7	183.6	62	91.2	226
1994	85	2.2	187.0	85	100.0	390
1995	91	2.0	182.0	91	100.0	501
1996	78	2.3	179.4	78	100.0	394
1997	76	2.4	182.4	75	98.7	284
1998	45	3.9	175.5	42	93.3	227
Mean	65.6	2.6	180.9	568	95.1	2423

[1]An additional 368 pigs were killed between 1990 and 1998, but these were on recreational or feral sheep hunts where no data were collected.

Development and evaluation of the base model

Age-specific survival and fecundity rates for the base model were derived from the kill data. We developed a vertical life table (Skalski *et al.* 2005) for each sex in each calendar year from 1991 through 1998, as well as a table for data pooled across years. The number of pigs killed in each age class x for each sex (N_{xi} where $i = m$ for males, f for females) was multiplied by the age-specific kill rate then subtracted from the total number killed (N) to obtain an estimate of the number alive in each age class (N_{xi}). Because we could not count the number of newborn pigs (age x = 0), we estimated the initial population size for each sex N_{0i} as

$$N_{0i} = \left(\sum_{x=1}^{n} N_{\dot{x}}\right) + \left[\left(\sum_{x=1}^{n} N_{\dot{x}}\right)(M_i)\right]$$

where M_i was the mean per capita litter size (m_x) in that year. The proportion of each sex alive at the start of each age interval (l_{xi}) was derived from the N_{xi}, and the age -specific survival rates (s_{xi}) were calculated from the l_{xi} values. Age-specific fecundities (f_{xi}) were calculated from the estimates of m_x and s_{xi}. Pigs breed year round on SCI, therefore estimates of l_{xi}, s_{xi}, and f_{xi} were calculated as birth-flow values (Caswell 2001).

We used a generalised linear model (GLM) with a binomial error structure and logit link to analyse the degree to which l_x values varied among years

$$\text{logit}\, l_x = \beta_0\, v_0 + u_{lx}\, v_{1ix}$$

where v_0 is a constant, v_{1ix} is the ith age class in the xth year, β_0 is an estimated parameter, and u_{lx} is an estimated parameter allowing l_x to vary randomly among years.

The estimates of s_{xi}, and f_{xi} were used to parameterise a two-sex Leslie matrix model M (Skalski *et al.* 2005) with nine age classes. Both sexes were included in the model because males and females of all ages would be harvested in an eradication programme. We assumed that a small number of pigs was originally introduced to SCI, therefore we used an initial vector N of 25 animals as the starting population size. We incorporated demographic stochasticity into the model by deriving a standard deviation matrix S from the observed temporal variation in s_{xi}, and f_{xi}. For each run of the model, values for s_{xi}, and f_{xi} were drawn randomly from a lognormal distribution based on their age-specific mean and SD. Based on observations of pig behaviour during a population crash (see RESULTS), we selected contest density-dependence as the form most likely representative of that on the island.

Carrying capacity (K) was estimated directly by regression of the rate of population change (λ) against estimated abundance in the prior year (N_{t-1}). We used a simple exponential equation

$$\lambda = c^*(\exp^{(-b^*N_{t-1})})$$

where C. and b are estimated parameters for the intercept and slope, respectively. Carrying capacity was estimated as abundance where the regression line intersected λ = 1. Environmental stochasticity was incorporated into the model by: 1) randomly drawing estimates of K from a lognormal distribution with a coefficient of variation (CV) of 0.25; and 2) a catastrophic event every decade (approximately one generation). The estimate of the CV of K was based on variation in mast counts collected annually

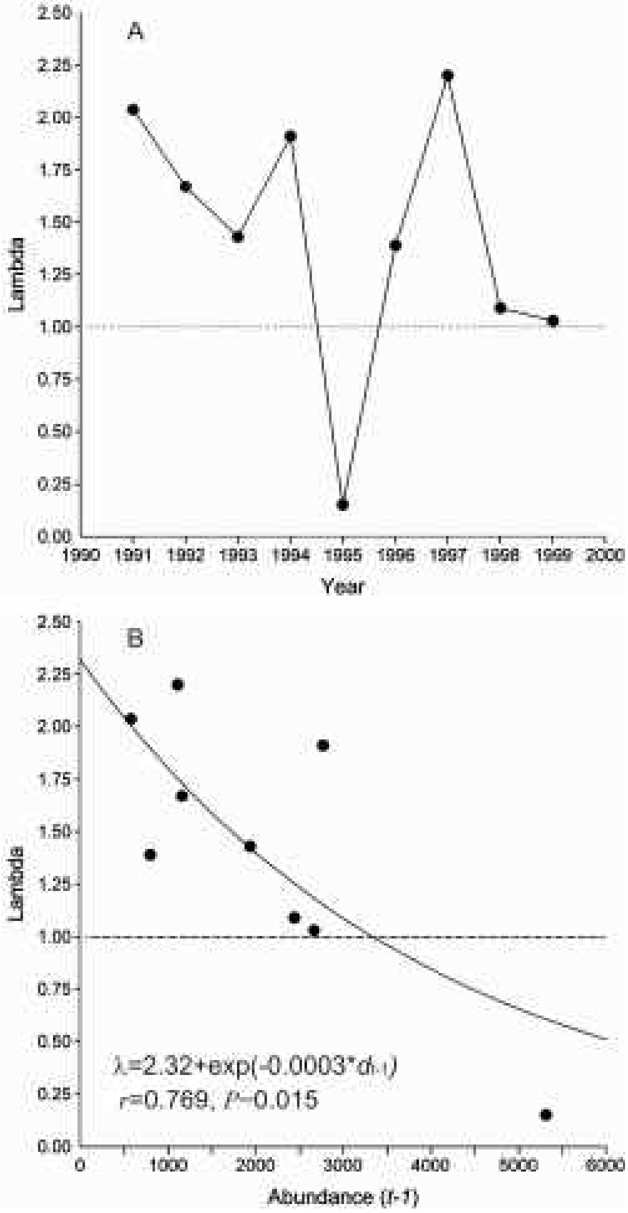

Fig. 1 (A) Variation in lambda (λ; the rate of population change) for feral pigs over a 10-year period on Santa Cruz Island, California; and, (B) the relationship between lambda and abundance of feral pigs in the previous year (d_{t-1}).

from 1990 through 1999 (R. Klinger, unpublished data), and the catastrophes represented mast failures, years of extreme drought, or both.

We evaluated performance of the models in three ways. First, we conducted 10,000 simulations (Caswell 2001) based on estimates of s_{xi}, and f_{xi} from each individual year (1991-98) and the model with years pooled (N = 9 models). We visually inspected the distribution of the mean population estimates in 5% percentile intervals for each model, and then compared the mean population estimates among them with standard least-squares ANOVA. Next, we used a jackknife procedure to derive estimates of s_{xi}, and f_{xi} by sequentially removing each year from the pooled model. We then conducted 10,000 Monte Carlo simulations for each model with a missing year, as well as the model with years pooled. We compared mean population estimates among the models with ANOVA and visual inspection of the distribution of the model estimates in percentiles at 5% intervals. All of the simulations spanned a period of 150 years. Finally, we generated 10,000 bootstrap samples

Table 3 Estimated abundance (± SE) of feral pigs on Santa Cruz Island, California, from 1990 through 2000. The model for all seasons is Half-normal w/ cosine and is the base model and expansion term used to derive the density estimate; Type is whether the data were analyzed ungrouped or grouped into predefined intervals; and N is the total number of observations used to derive the estimates. Distance data were collected along transects annually from late November through early March (wet season).

Season	Abundance	Type	N
Wet 1990-91	579 ± 97	Ungrouped	71
Wet 1991-92	1161 ± 130	Ungrouped	114
Wet 1992-93	1940 ± 300	Ungrouped	85
Wet 1993-94	2776 ± 428	Grouped	91
Wet 1994-95	5315 ± 984	Grouped	99
Wet 1995-96	801 ± 108	Grouped	83
Wet 1996-97	1110 ± 199	Grouped	78
Wet 1997-98	2444 ± 454	Ungrouped	57
Wet 1998-99	2670 ± 416	Ungrouped	112
Wet 1999-00	2753 ± 387	Ungrouped	88

consisting of 150 random draws of abundance and its CV from the models generated in the previous steps. We then determined which percentile of the bootstrapped values the mean estimates of abundance and CV for each individual model fell.

Harvest Models

We incorporated annual harvest rates (h) from 5% to 95% at 5% intervals for models with three different starting levels of island-wide abundance (N_0): a low abundance model where N_0 = 800 (approximately 25% of K; see RESULTS), a mean abundance model where N_0 = 2400 (75% of K;), and a high abundance model where N_0 = 5000 (150% of K). Harvest effort was targeted equally among all sex and age classes for ten years. To simplify interpretation of the trajectories we set the environment as constant (CV K =0) and eliminated catastrophes. We conducted 10,000 runs for each of the three models, then calculated the probability of eradication (Pr_e), the median time to eradication in years for $0 \leq Pr_e < 1$, the time to eradication for $Pr_e = 1$, and the mean percent reduction in abundance at each for the harvest rates in each model.

RESULTS

Abundance, population change, and carrying capacity

Abundance of feral pigs on SCI ranged from 579 (± 97 SE) in 1990/91 to 5315 (± 984 SE) in 1994/95 (Table 3). The coefficients of variation ranged from 11.1% to 18.9%. The population exhibited a "boom or bust" pattern, with a steady increase in abundance from 1990/91 through the wet season of 1994/95, followed by a severe crash the following year. The population recovered rapidly though, and continued to increase through the wet season of 1999/2000 (Fig. 1A). There was a significant negative relationship between lambda and N_{t-1} (r = 0.769, $F_{1,7}$ = 10.13, P = 0.015). With the exception of the wet season 1994/95, lambda tended to decrease as island-wide abundance of the pigs approached 3000 (Fig. 1B). Abundance for lambda = 1 was 3400, which was used as the estimate of K.

Population structure and evaluation of the base model

Sex and age data were collected for a total of 2423 pigs. The sex ratio of the population was approximately 1:1 (N = 1221 females, N = 1202 males). Values for l_x between 1991 and 1998 are given in Table 4. Model-derived estimates from the GLM analysis indicated that variation in l_x was similar across years (Fig. 2).

Table 4 The estimated proportion of feral pigs surviving at the start of nine age classes (years) on Santa Cruz Island, California.

Age Class	1991	1992	1993	1994	1995	1996	1997	1998	Years Pooled
1	0.553	0.427	0.394	0.537	0.490	0.575	0.430	0.522	0.489
2	0.381	0.320	0.270	0.396	0.375	0.429	0.326	0.359	0.357
3	0.279	0.192	0.155	0.219	0.247	0.315	0.230	0.223	0.231
4	0.195	0.068	0.044	0.067	0.074	0.109	0.070	0.101	0.072
5	0.106	0.039	0.019	0.032	0.028	0.039	0.024	0.041	0.031
6	0.053	0.014	0.003	0.007	0.004	0.007	0.006	0.005	0.008
7	0.004	0.009	0.000	0.001	0.001	0.000	0.005	0.002	0.002
8	0.000	0.002	0.000	0.000	0.000	0.000	0.002	0.000	0.000

There were no simulated model runs where the population went naturally to extinction. The mean minimum estimate of abundance from the models based on individual years was 805 ± 23 SE and from jackknifed models 847 ± 8 SE. Mean maximum abundance from models based on individual years was 6257 ± 64 SE and from jackknifed models 6314 ± 63 SE. The range in percentile abundance among years for both individual and jackknifed models was 17.6%, with 90% of the mean estimates of annual abundance falling between 1200 and 5900 (Table 5). Although the relative range among the simulated estimates tended to be < 20%, the greatest differences were in the 5th percentile. All mean abundance and CV abundance estimates from the individual models fell within the 32nd and 71st bootstrap percentiles. There was no significant difference in mean estimates of simulated feral pig abundances for models based on simulations from individual years ($F_{8,1350}$ = 1.433, P = 0.178) or the jackknifed models ($F_{8,1350}$ = 0.641, P = 0.744).

Table 5 Estimated percentiles of abundance from two groups of models simulating feral pig abundance on Santa Cruz Island, California. Individual models were simulations run separately for each year, as well as an additional one with years pooled. Jackknifed models were run with one year removed from each simulation. Each simulation consisted of 10,000 runs over a 150-year period.

Individual Models	Percentile				
	5th	25th	50th	75th	95th
1991	1641	2648	3088	3855	5018
1992	1643	2596	3061	3945	5230
1993	1583	2278	2869	3912	5892
1994	1621	2371	3089	3793	5460
1995	1640	2503	3378	4038	5023
1996	1501	2554	3145	3808	5214
1997	1466	2526	3117	3752	5052
1998	1200	2262	2907	3693	5699
Pooled	1252	2618	3424	4027	5337
Mean	1505	2484	3120	3869	5325
Range	443	386	555	345	874
Range (%)	29.4	15.5	17.8	8.9	16.4
Jackknifed Models	5th	25th	50th	75th	95th
No 91	1293	2363	2980	3773	5687
No 92	1226	2065	2867	3501	5648
No 93	1518	2073	2603	3897	5683
No 94	1577	2255	3046	3790	5446
No 95	1333	2029	2850	3440	5404
No 96	1411	2372	2965	3672	5537
No 97	1383	2070	2824	3822	5671
No 98	1402	2225	3027	3604	5637
Pooled	1388	2429	3054	3799	5123
Mean	1392	2209	2913	3700	5537
Range	351	401	451	457	565
Range (%)	25.2	18.2	15.5	12.4	10.2

Because there was little evidence of systematic differences among the models, we selected the base model to be the one with demographic rates derived from the years pooled together. The mean and CV of the simulated abundance estimates from the pooled model were well within the range of bootstrapped estimates, and deriving estimates of s_{xi}, and f_{xi} from kill data collected across years was likely the most appropriate approach for integrating the observed temporal variability in vital rates into the simulations. The mean minimum and maximum abundance from 10,000 simulated 150-year time series of the base model were 669 (±107 SE) and 5645 (±636 SE), respectively. The mean value of λ was 1.118 ± 0.128 SE. The population did not reach zero in any of the simulations for the base model.

Harvest Models

The effects of increasing harvest rates (h) on pig abundance for the three initial levels of abundance are shown in Fig. 3. At low initial abundance (N_0 = 800) h > 45% was required to reduce abundance below N_0, and h > 60% was required to prevent the population from becoming stable. Levels of abundance for 45% < $h \leq$ 60% were 40-80% below N_0 = 800. Harvest rates ≥ 20% initiated declines when N_0 = 2400, but h > 45% was required to keep the population from stabilising. Levels of abundance for 45% < $h \leq$ 60% were 10-87% below N_0 = 2400. Severe declines in pig abundance were independent of harvest when N_0 =

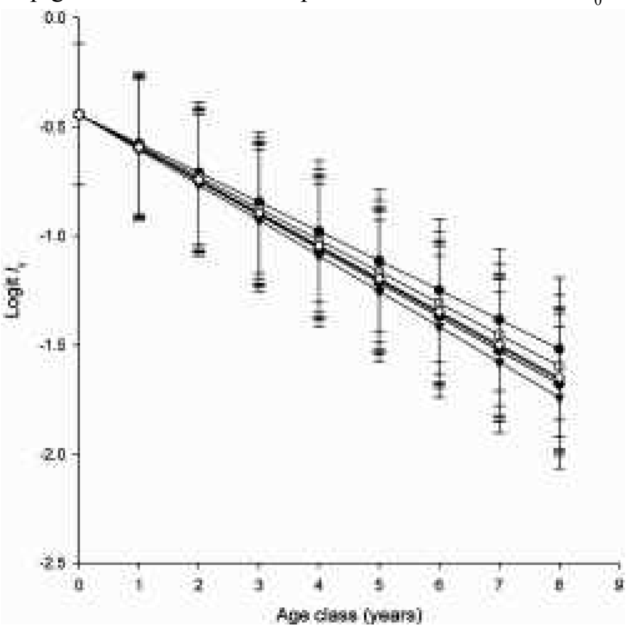

Fig. 2 Estimated variation in the proportions of feral pigs surviving at the beginning of eight age classes (l_x on a logit scale) from 1991-1998 on Santa Cruz Island, California. Year was modelled as a random effect, with each line representing the l_x distribution in any given year.

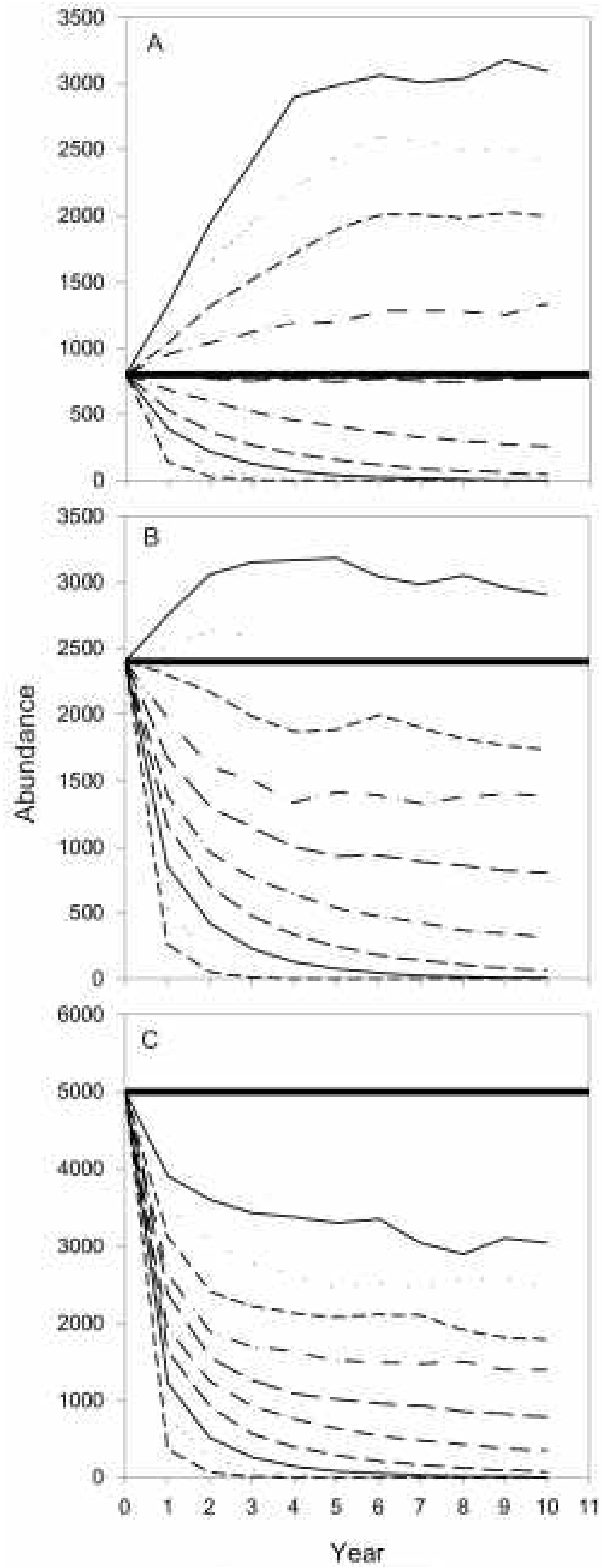

Fig. 3 Results of simulated harvest rates on a feral pig population on Santa Cruz Island, California, over a 10-year period. There were three starting levels of abundance (heavy black horizontal limit lines); N_0 = 800 (A), N_0 = 2400 (B), and N_0 = 5000 (C). Harvest rates range sequentially from 0 (solid line) to 90% in 10% increments.

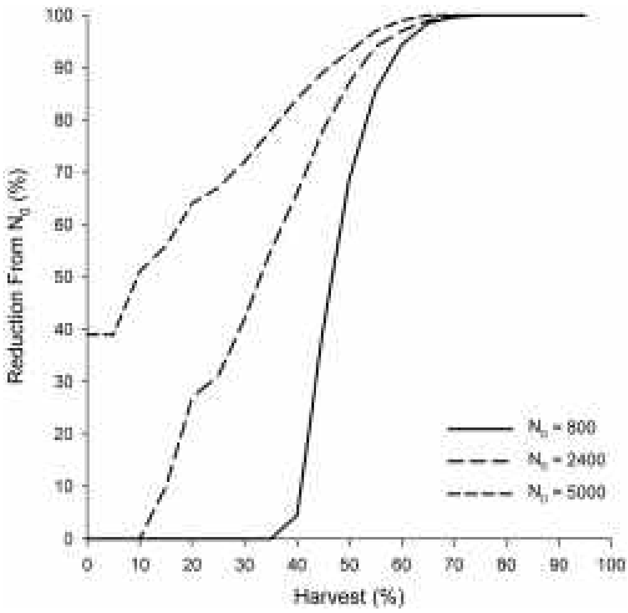

Fig. 4 Simulated rate of reduction at different harvest rates for a feral pig population on Santa Cruz Island, California. Simulations (N = 10,000) were run for three levels of initial abundance (N_0).

5000. Modest harvest rates of 10-35% during the decline phase when N_0 = 5000 reduced abundance to stable levels; levels of abundance for 10% < $h \leq$ 35% were 26-68% below K. The population at N_0 = 5000 continued to decline when h > 35% (Fig. 3). The initial size of the population had a strong influence on proportional reduction relative to N_0 at low to moderate harvest rates (5%-50%), but the influence decreased as harvest rates approached 70% (Fig. 4). By year 10 of the simulations, the 95% confidence intervals for all three initial population sizes overlapped that of the unharvested population when h < 10%.

There was no probability of eradication until h > 70% (Fig. 5). The probability of eradication (Pr_e) was < 1 for 70% < h < 80% (Fig. 5), but as h approached 80% Pr_e rapidly increased. For h = 70% values of Pr_e ranged from 0.02 to 0.09, but when h > 75% values of Pr_e ranged from 0.97 to 0.98. Pr_e = 1 when h > 80%. There was

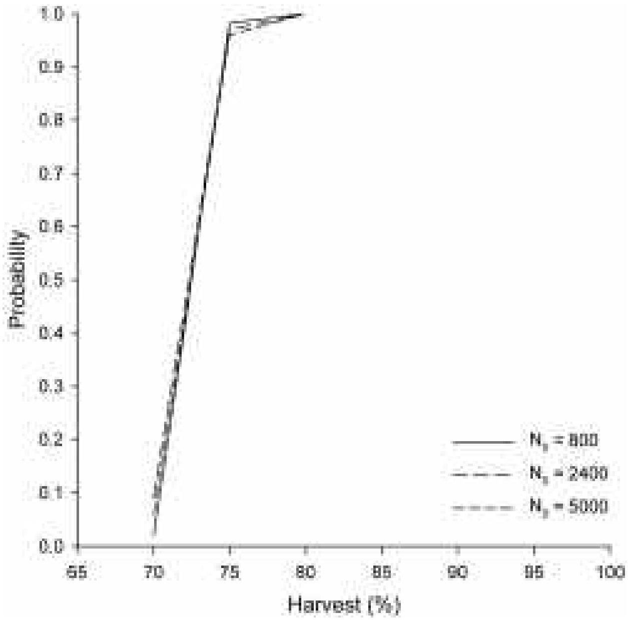

Fig. 5 Simulated probability of eradication at different annual harvest rates for a feral pig population on Santa Cruz Island, California. Simulations (N = 10,000) were run for three levels of initial abundance (N_0).

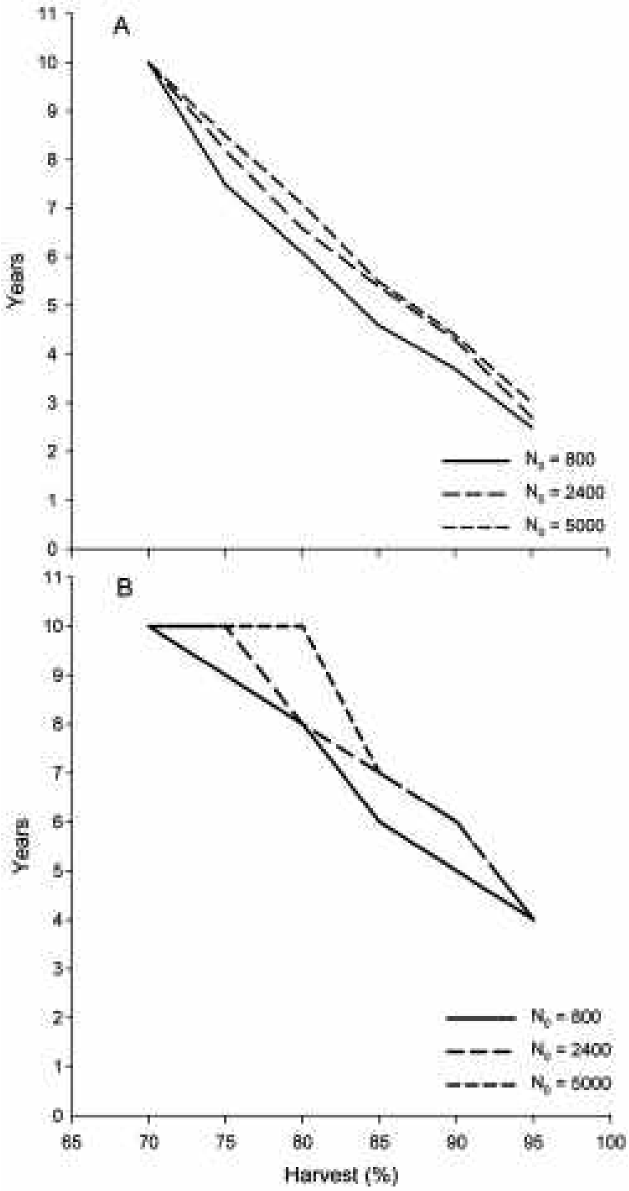

Fig. 6 Simulated time to eradication at different annual harvest rates for feral pigs on Santa Cruz Island, California. Simulations (N = 10000) were run for three levels of initial abundance (N_0). Panel (A) is based on the median number of years when probability of eradication (Pr_e) < 1. Panel (B) is based on the number of years when Pr_e = 1.

no relationship between N_0 and Pr_e (Fig. 5), but N_0 did influence the number of years to eradication (Fig. 6). The median number of years to eradication ranged from ten (72% annual harvest rate) to 2.5 (95% annual harvest rate). There was a linear decrease in median years to eradication for all three levels of abundance (Fig. 6). Median years to eradication for programmes initiated when N_0 = 800 was predicted to be between 3 and 9 months less than those begun when N_0 = 2500. Programmes initiated when N_0 = 800 were predicted to be between 6 and 12 months shorter in duration than those begun when N_0 = 5000. Eradication programmes that began when N_0 = 2500 were predicted to be completed 1-6 months sooner than those initiated when N_0 = 5000 (Fig. 6).

Time to eradication when Pr_e = 1 decreased linearly with increasing rates of annual harvest for N_0 = 800. Time to eradication also decreased linearly when h > 75% for N_0 = 2400, while the pattern of decrease for N_0 = 5000 exhibited a more stepwise pattern (Fig. 6). Eradication programmes that were initiated when N_0 = 800 were generally a year shorter in duration than those that began when N_0 = 2400 for 70% < h < 95%, and 1-2 years shorter than those that began when N_0 = 5000 for 70% < h < 95% (Fig. 6). Time to eradication when Pr_e = 1 for N_0 = 2400 and N_0 = 5000 were the same at all harvest rates except h = 80%, which was the threshold value for N_0 = 5000 (Fig. 6).

DISCUSSION

Simulations of the effect of varying harvest rates on abundance of feral pigs modelled when a population was likely to be controlled and when it was likely to be eradicated. For example, attempts to manage pig populations with annual harvest rates below 10%, which are likely typical of sport hunting, will have little or no detectable effect on abundance (Barrett *et al.* 1988; Waithman *et al.* 1999). Harvest rates in the range of 15% to 50% will reduce and maintain numbers below that of a population that is not hunted, but abundance may still be greater than desirable relative to conservation goals. For instance, in models with moderate and high levels of initial abundance (N_0 = 2400 and N_0 = 5000), annual harvest rates below 45% resulted in population size in excess of 1000 individuals even after 10 years of hunting. When actual numbers of pigs were above this level on SCI, they continued to have undesirable ecosystem and species-specific effects, including widespread rooting and impacts to two species of rare endemic plants (Klinger *et al.* 2002; Klinger 2007). So, while pig numbers can be controlled with annual harvest rates between 15% and 50%, their reduced abundance may still be above that required to meet conservation objectives.

Mortality from hunting was largely compensatory at low annual harvest rates (5%), but became additive as rates increased beyond 10%. However, the importance of the additive mortality depended on harvest rates and the abundance of the population when hunting commenced. At low abundance, the rate of growth was high enough that, despite mortality being additive, control was unlikely if the annual harvest rate was between 5% and 40%. When initial population size was low, and annual harvest rates were between 45% and 65%, control became more likely. When initial population size was relatively high, but still below carrying capacity, control was likely when annual harvest rates exceeded 20%. This likely reflected the additive effects of harvest and the influence of negative density-dependence. Not surprisingly, when abundance exceeded carrying capacity strong negative density-dependence resulted in rapid population declines. Initiating harvest as the population declined pushed it to lower abundance than from density-dependent processes alone. When annual harvest rates were between 10% and 60%, the population still stabilised, albeit at progressively lower abundance. There was little likelihood of eradication unless annual harvest rates exceeded 75% per year. However, when harvest rates exceeded 75% then additive mortality had a very significant influence on the population and the likelihood of achieving eradication became independent of the initial level of abundance.

Although there was little possibility of eradication until annual harvest rates were greater than 75%, harvest rates between 60% and 70% reduced the population sufficiently for the pigs to be considered ecologically extinct. This condition would likely be acceptable if the goal of the management programme was control rather than eradication and there were financial resources available to sustain hunting. In this case, the effects of pigs as a transformer species would be eliminated and there would be far less likelihood of impact to high value species, such as rare endemic plants (Klinger *et al.* 2002). However, maintaining low numbers as a long term conservation strategy could be very risky. Animal removal programmes are controversial, so sustaining institutional support and financial resources for long-term control may be unrealistic when faced with strong public opposition (Sagoff 2005; Perry and Perry 2008). Moreover, the expenditure of resources would be much greater to reduce and then maintain a population at low levels rather than implement a relatively short term but intense eradication programme (Cruz *et al*. 2005, 2007). These possibilities could result in situations that would be considered "a conservation nightmare"; that is, the cessation of control and the subsequent return of the population to previous levels of abundance (Campbell and Donlan 2005).

While the predicted ranges in abundance among the models tended to be relatively consistent, the results should still be interpreted with caution. Estimates of fecundity and survival derived from vertical life tables can be biased if data are collected from a single sample of a population when growth rates are not constant (Caughley 1977). Rates of change in the pig population were clearly not constant during the study, but our estimates of sex and age structure were collected across multiple years. This likely reduced error in the estimates, but some degree of bias is still possible (Caughley 1977; Skalski *et al*. 2005).

Comparison of the simulations with actual eradication programmes suggests that the estimates for time to eradication are realistic, though in some cases they may be somewhat conservative. For example, more than 18,000 feral pigs were removed from Santiago Island (Galapagos Islands; 584 km^2) over a 30-year period, but the first phase of this project was largely a low-intensity effort with little evidence of substantial control (Cruz *et al*. 2005). When rates of removal were increased in 1995, the remaining few hundred pigs were eradicated within six years (Cruz *et al*. 2005). A similar pattern was reported from Santa Catalina in the Channel Islands (194 km^2), where more than 12,000 pigs were removed from 1990-2003 (Garcelon *et al*. 2005). For the first seven years, the focus on Santa Catalina was control, but when it became an eradication programme in 1996, 2679 pigs were removed within seven years (Schuyler *et al*. 2002, Garcelon *et al*. 2005). Eradication of 200 pigs from a 57 km^2 fenced area at Pinnacles National Monument in central California, USA, was completed in 2.5 years (McCaan and Garcelon 2007), and 1206 pigs were eradicated from Santa Rosa Island (Channel Islands, California, USA; 215 km^2) in three years (Lombardo and Faulkner 2000). Eradication of 143 pigs from Annadel State Park (20 km^2) in central California was accomplished in under three years (Barrett *et al*. 1988).

Other cases suggest that eradication times can be substantially reduced from those predicted by the models. One factor is the size of the eradication area; eradication in very small areas with low pig density can be accomplished in a year or less (Kessler 2002). More important factors, though, may be a combination of resource allocation, technology, and hunting techniques, especially in larger areas. When eradication of feral pigs was undertaken on SCI, NPS and TNC invested considerable funds in fencing, helicopters, large numbers of hunters and dogs, Judas animals, strategically and tactically integrated hunting techniques, GIS and GPS technologies, and systematic monitoring (Morrison *et al*. 2007). These factors, as well as the commitment by NPS and TNC to eradicate and not control the population, resulted in the removal of 5036 pigs in 15 months, approximately 5-10 years less than anticipated (NPS 2003; Parkes *et al*. 2010).

The results of the simulations are likely applicable to many insular systems, but they may be less applicable to mainland systems where pigs have more predators and competitors (Barrett 1978). Competition between pigs and other vertebrates is rarely reported, and when it does exist it may alter patterns of distribution rather than reduce abundance (Ilse and Hellgren 1995). Predation could lead to significantly different estimates of vital rates though, especially survival (Woodall 1983, Okarma *et al*. 1995). Moreover, dispersal from areas where pigs are not being controlled is likely to act as a "rescue effect" for sink populations where control efforts are underway (Barrett *et al*. 1988). Indeed, in many areas, eradication may not be a feasible option unless expensive measures are taken to prevent recolonisation (Hone *et al*. 1980; Barrett *et al*. 1988; Garcelon *et al*. 2005; McCaan and Garcelon 2007). In situations where such measures (eg. fencing) cannot be used, there may be few options other than control. At that point, a key decision will be what long term harvest rates can be sustained to prevent pigs from becoming too abundant (Cowled *et al*. 2006).

MANAGEMENT IMPLICATIONS

Our models suggest that, in general, a strategy of intense harvest for five years will likely eradicate many insular feral pig populations. When options are limited to some form of control, development of population models would be a substantial aid in justifying target harvest rates and developing monitoring programmes to evaluate if conservation goals are being met. But even when institutions are willing to commit fully to eradication, investing in the collection of several years of data to develop models projecting the likelihood of eradication for different harvest scenarios would help with planning and design.

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Eradication of beaver (*Castor canadensis*), an ecosystem engineer and threat to southern Patagonia

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ABSTRACT Beavers threaten ecosystems of global importance in southern Patagonia, causing significant impacts on biodiversity. Introduced in 1946, they have reached all the most important islands south of the Strait of Magellan and now are invading the Brünswick Peninsula, on the South American continent, occupying a total area of approximately 70,000 km^2. After years of trying to promote beaver control by stimulating their commercial exploitation, the governments of Argentina and Chile agreed to redirect management efforts and to attempt to eradicate the species throughout its entire range in South America. As a first bi-national activity, a feasibility study, conducted by international experts, was jointly initiated, to assess the technical, ecological, economic, social and cultural feasibility of eradicating beavers over their entire range. This study indicated that eradication was justified and feasible, although several issues must be resolved before an eradication operation is implemented. Beaver eradication in southern South America presents unique challenges, as well as unique opportunities, to develop a new cooperative model to handle complex and global environmental problems.

Keywords: American beaver, southern Patagonia ecosystems, new approach, bi-national agreement, Chile, Argentina

INTRODUCTION

Beavers (*Castor canadensis*) are ecosystem engineers that directly or indirectly control the availability of resources for other organisms by causing changes in the physical state of ecosystems (Jones *et al*. 1994). Exotic in South America, beavers are threatening biodiversity values of global significance in southern Patagonia, where temperate forest and peat bog ecosystems dominate the landscape in one of the world's largest and most pristine remaining wilderness areas. Together, these ecosystems play a key role in global circulation processes, since they constitute the most significant terrestrial carbon reservoirs and carbon sinks in these latitudes. However, Subantarctic ecosystems appear to be particularly vulnerable to invasion by introduced species (Mittermeier *et al*. 2001) such as beavers, which now impact the largest stands of Subantartic forests and Holocene peat bogs. This invasion is a good example of how the human footprint can dramatically reach the last of the wild areas of the world, and how global hazards, like biological invasions, can affect biodiversity and key ecological processes in very remote areas.

In order to establish a new fur industry, 25 breeding pairs of North American beaver were introduced in 1946 to Río Claro´s lower basin, south of Tierra del Fuego Main Island, the largest island of the Fuegian Archipelago (48,000 km^2) (Fig. 1). This archipelago, at the southernmost tip of South America, consists of hundreds of islands administered by Chile and Argentina. The area is surrounded by the Atlantic and Pacific Oceans and is influenced by an Antarctic climate, with extreme cold and wet conditions. In southern Patagonia, beavers have flourished with abundant food, water, and a virtual lack of predators and competitors. This has favoured their expansion, allowing them to colonise all existing habitats including deciduous and evergreen beech forests, peat bogs, Patagonian steppe, and Andean grasslands (Saavedra and Silva 2008). The beavers have since spread throughout the entire Fuegian archipelago and beyond.

The rate of beaver expansion has been estimated at 2-6 linear km/year (Lizarralde *et al*. 1996), and the total population is about 60,000 individuals (Skewes *et al*. 1999). In the first twenty years after their introduction, beavers occupied about 30% of the rivers of the Andean zone of the Main Island of Tierra del Fuego (Lizarralde 1993) and were recorded in Chilean territory in the 1960s, 16 years after their release in Argentina (Lizarralde 1993; Lizarralde and Escobar 2000; Lizarralde *et al*. 2004). Beavers subsequently crossed the Beagle Channel in 1962 and colonised the northern coast of Navarino Island. They are now found on almost all of the islands south of the Strait of Magellan, including the entire Isla Grande of Tierra del Fuego, Picton, Lenox, Nueva, Hoste, and Dawson (Fig. 1). In their invaded range, beavers have affected over 20,000 linear kilometres of streams, rivers and watersheds and their density is estimated in 0.7 colonies/km^2. In the 1990s, beavers crossed the Strait of Magellan and established on the Brunswick Peninsula, where they are starting to invade the southernmost part of the South American continent (Soto and Cabello 2007) (Fig. 1). The total area occupied is now estimated as 70,000 km^2.

Beavers in southern Patagonia have had significant impacts on native species, habitats, ecosystems and landscapes (Figs. 2A and B). *Nothofagus* forests have been particularly affected with understory diversity, structure and natural dynamics impacted in the cut and flooded zones and in abandoned ponds (Anderson *et al*. 2006). Beaver

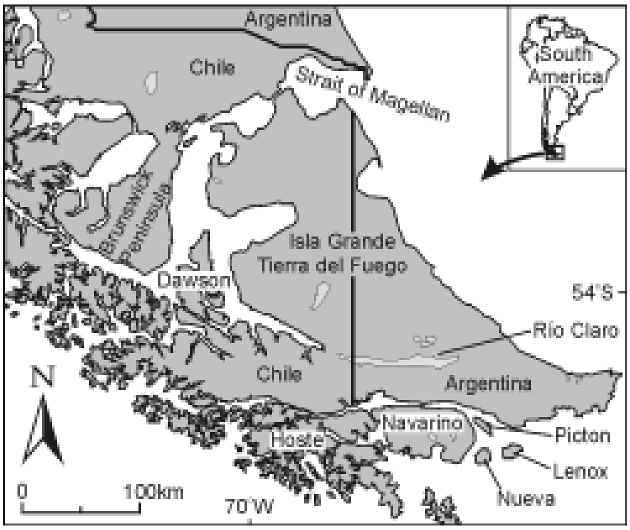

Fig. 1 Tierra del Fuego archipelago and Brunswick Peninsula in the South American continent where the introduced beaver population is spreading.

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dams, which directly change hydrological processes, have caused other serious impacts and sediment flows. Dams transform lotic environments into lentic ones. By creating flooded areas, beavers change drainage patterns and water table depth, cause the accumulation of sediments and organic matter, facilitate the alteration of nutrient cycles in *Nothofagus* forests (Lizarralde *et al*. 1996; Jaksic 1998; Lizarralde *et al*. 2004), accelerate the decomposition process, and alter water and soil chemistry (Lizarralde *et al*. 1996), with consequences to benthic and vertebrate communities (Anderson *et al*. 2009). Perhaps the most obvious impact of beaver invasion is the direct destruction of riparian southern beech forests preventing the natural recovery of forest ecosystems, which in the long term are transformed into grassland (Anderson *et al* 2006; Martinez Pastur *et al*. 2006).

Beavers also have economic impacts affecting aquaculture, agriculture and particularly forestry. These are important local industries and support a significant portion of the Chilean and Argentinean economies. Flooding as a result of beaver activity reduces the availability and quality of pastures for livestock, blocks culverts and destroys bridges and roads.

While they were confined to Tierra del Fuego, beavers were a remote problem. After crossing the Strait of Magellan and reaching the Brunswick Peninsula, beavers are now recognised as a serious threat to biodiversity and the economy of southern South America. The northward expansion of beavers, which is inevitable unless their expansion and establishment is stopped, will destroy further forests and lead to greater watershed contamination.

Fig. 2 Ecological impacts produced by exotic beavers in Tierra del Fuego ecosystems. A Vestiges of an original beach forest replaced by a "Beaver meadow" in and abandoned site. B Beaver lodge in a dammed and flooded riverine environment.

In this paper we describe a new bi-national approach that is being developed to manage beaver populations in Southern Patagonia. This approach will strategically test if the shift from localised control to eradication of all populations is achievable.

BEAVER CONTROL IN SOUTHERN PATAGONIA

In Argentina, control of beavers started in 1981 when the government authorised recreational hunting, followed by authorised commercial hunting in 1997 and the first management plan in 1999. At that time, the beaver populations was estimated at 30,000-50,000, increasing at 0.21- 0.23 and was close to maximum capacity (Lizarralde 1993). An elimination rate was set at 21-23% per year, with a required extraction of 7000-10,000 animals in the same period. Control was based on sustainable culling, implemented by local trappers, using Conibear 330 traps and assumed the creation and maintenance of a market for beaver products. Since 2001, a bounty was also paid for every tail delivered to local authorities. Together with the development of a fur market, the bounty was intended to provide an additional stimulus for beaver trapping. Furthermore, in order to stimulate beaver trapping, use of the meat was promoted. Despite these efforts, the necessary and planned extraction rate was not achieved. The Government also failed to maintain a monitoring system to guide or improve management decisions.

In Chile, the National Agriculture and Livestock Service officially recognised beavers as a harmful species in 1992. The first control programme was implemented in 1999, focused on the Isla Grande of Tierra del Fuego and Navarino Island. As in Argentina, the goal was to promote the economic benefits of the species and included a bounty system for private trappers who could profit from beaver pelts and meat. In 2004-2006, the Chilean government continued this programme, reinforcing the bounty system to promote beaver capture and the creation of a market for beaver products and sub-products. As a tool to mitigate beaver impacts, beaver hunting was concentrated in those areas closer to the mainland. As in Argentina, these plans failed to create a market, to promote beaver trapping, or to limit expansion of the beaver's range.

A KEY CHANGE OF VISION: BEAVER ERADICATION IN SOUTHERN PATAGONIA

Because efforts to control beavers were insufficient to reduce the beaver population or limit its expansion, beavers crossed the Strait of Magellan and established on the South American continent. In response, control programmes were critically reviewed and an historic first bi-national scientific and administrative agreement was reached in 2006 by the governments of Argentina and Chile. Both countries agreed to cooperatively work towards eradicating beavers throughout southern Patagonia. Key stakeholders from Chile and Argentina, along with international advisors, have since been working on ways to implement this new strategy. The goal is to restore natural southern Patagonian ecosystems through the eradication of beavers from their entire non-native range.

Key steps in this new approach include:

- A feasibility study conducted by a team of international experts and financed by the Governments of Chile and Argentina and the Wildlife Conservation Society.
- A bi-national agreement signed between Argentina and Chile (2008) under the bi-national Treaty on Environment (1992) and the specific shared Wildlife Protocol, to work towards beaver eradication as a necessary step to restore southern Patagonian ecosystems.
- The preparation of a Strategic Plan for the eradication of beavers from southern Patagonia, which will be adopted as a bi-national strategic reference document.

The feasibility study

This study assessed the technical, ecological, economic, social and cultural feasibility of beaver eradication over their entire range, whether beaver eradication was possible in a regional context, and if it is justified in terms of potential benefits relative to costs.

Different management options were: 1) removing beavers currently on the mainland of South America, with sustained control of source populations in buffer zones in the Fuegian Archipelago, along with surveillance and rapid response at mainland sites; 2) eradicating beavers from Tierra del Fuego and the South American continent; and 3) sustained control of beavers and other invasive species in high priority areas.

The eradication of beavers from southern Patagonia would avoid increasing damage on the continent and would remove impacts on biodiversity and economic values within the beavers' current range. Eradication would be a preliminary step to restoring southern ecosystems.

The control options would require the perpetual removal of beavers in specific areas in order to maintain impacts within acceptable levels. These options involve the sustained and regular input of resources that should be allocated to specific sites, which are selected and prioritised for their conservation values, or the need for protection from harm (Parkes *et al*. 2008).

Although eradication is ecologically, technically, and economically feasible, constraints include: 1) access to all types of land tenures (e.g., military lands, private lands) must be guaranteed; 2) organisational complexities involved with the bi-national character of the project must be resolved; 3) capacity to implement the project is currently absent from the region and must be developed; 4) technical and logistical complexity due to isolation and weather will need to be overcome; and 5) other minor constrains derived from the presence of native species that could incidentally become targets (e.g., native otter *Lontra provocax*). Such constraints present risks of failure that need to be tested, and management responses must be resolved before any eradication operation proceeds (Parkes *et al* 2008).

The feasibility study also identified risks and limitations and raised key questions that need to be answered before or during the implementation of each strategy, along with an indication of necessary resources and possible actions required (e.g., research, demonstration or pilot projects, monitoring). Also, it was clearly established that all technical, political, legal and operational tools must be available to guarantee complete beaver removal before any active eradication operation is started.

The removal of beavers from the continent was identified as of high priority and urgency. Its goal is to maintain areas at "zero density", which implies a permanent/sustained capture and monitoring regime to reduce immigration and reinvasion of beavers in the managed area. Beaver colonisation on the mainland seems to be slower compared to the island of Tierra del Fuego. Different invasion rates could be explained by reduced propagule pressure, or by the presence of predators such as pumas (*Felis concolor patagonica*) that are absent from islands (Wallen *et al*. 2007; Parkes *et al*. 2008). All these hypotheses remain to be tested.

All of the management options require assessment of the geographical range of beaver populations to ensure that all individuals in targeted populations are removed. Moreover, among other issues, the following additional information will be required for an effective management strategy: 1) mechanisms of beaver migration and establishment at continental sites; routes or pathways for access and movement on the continent; 3) frequency of immigration pulses; 4) sources of beaver populations: and 5) the relationship between dispersal and density of beavers.

Detection and surveillance methods that use probabilistic methods as a tool to provide transparent decision-making will be needed for areas to be declared beaver-free, and also to ensure the quick detection of any new arrivals.

The feasibility study recommended that pilot or demonstration projects should be used to resolve some of the above issues and to evaluate operational aspects of the eradication (Parkes *et al*. 2008). These projects should address key research, training and capacity-building objectives at different levels (e.g., public agencies, trappers, scientists). An adaptive process will also be required in order to learn and build capacity. The ultimate goal of this process will be to generate best practice and the highest operating standards to be applied in the effective planning, implementation and monitoring of a beaver eradication operation. Pilot or demonstration projects could also be used to present approaches and advances to key stakeholders such as politicians, financiers and other important actors needed to support and strengthen any eradication programme.

We suggest that the beaver eradication project should be organised in phases. Phase one should include establishment of the project and declarations of support from management agencies of both countries, as well as from other national and international stakeholders. It will also include the development of necessary capacities within management agencies to fulfil their roles and complete tasks to agreed standards. This will involve training in such varied fields as communications, population modelling and using radio transmitters. Project governance policies and procedures will need to be established. Baseline monitoring will need to be initiated and relevant management-driven research undertaken.

The eradication operation will involve beaver removal, beginning zone by zone, following a tactical, systematic, and adaptive approach.

The last phase of the project will involve monitoring and on-going surveillance.

Although no deliberate beaver eradication project has been undertaken previously -many populations of *Castor canadensis* were destroyed or heavily reduced by commercial fur trapping pressure in vast areas of their original range in North America (Baker and Hill 2003). Succession processes after pond abandonment and the effects that influence this process have been widely studied (Naiman *et al.* 1994, Collen and Gibson 2000, Wright *et al.* 2003, Anderson *et al* 2009, Burchsted *et al.* 2010, Hay 2010). However, it is unclear how this research will apply in South America. It thus remains unknown whether beaver removal, by itself, would be enough to promote the recovery of the ecosystems to a pre-beaver condition, at least in the short term. There is evidence (Martínez Pastur 2006) that *Nothofagus* forest restoration could need to be re-enforced by other practices such as long-term commitments to ecosystem management at the watershed level (Anderson *et al.* 2006)

Since beaver removal is aimed at the restoration of Patagonian ecosystems, specific information on restoration must be developed along with the eradication implementation, to assess the capacity and speed of recovery of ecosystems. Implementing appropriate measures to mitigate potential impacts of eradication activities should move the system into more acceptable trajectories (Parkes and Panetta 2009).

The cost of beaver eradication, which includes preparation, undertaking the operation and early stages of surveillance, but excluding on-going monitoring, is estimated at about US$ 35 million (Parkes *et al*. 2008). Although only indicative, the estimate includes the major cost components such as staff, equipment, and logistics.

Helicopters were viewed as an essential tool to implement the eradication, due to the large areas involved, their inaccessibility, the need to work in the shortest time possible to minimise risk of recolonisation, and to maintain commitment to the project at all levels (e.g., operational, governments, funding agencies). Helicopters are not widely used for conservation purposes in Argentina and Chile, although there is experience in their use for forest fire control activities and spraying crops.

Project governance will be challenging because it involves bi-national collaboration over administration, making decisions and evaluating progress. This project also has additional complexities including a large spatial scale, relatively long duration, logistical difficulties, and political, social and cultural challenges due to its bi-national nature, with the derived involvement of multiple jurisdictions, entities and organisations. These high levels of complexity will require the development and implementation of an appropriate governance structure and procedures to achieve project objectives (Parkes *et al*. 2008). Good governance also entails explicit processes for decision-making, and the establishments of transparent and efficient processes with clear lines of accountability. Moreover, appropriate and effective governance will be the key to retaining the political support required for the project to be implemented and for project goals to be achieved.

CONCLUSIONS AND NEXT STEPS

The processes so far completed have already provided useful lessons. First, beaver invasion in southern Patagonia is a global as well as a local problem. Addressing them in southern Patagonia will require international as well as local and national inputs. Second, the beaver problem must be completely understood by stakeholder communities, as well as by government authorities in Argentina and Chile (Soto *et al*. 2008). Third, beaver eradication in southern Patagonia appears feasible, but will be an enormous challenge. Finally, the environmental and economic benefits from beaver eradication will be extraordinary, and therefore, it is worth the effort to try to eradicate them.

Decision-making and the implementation of operational plans will now be guided by the strategic plan, which will provide an important basis for preparing funding proposals, and for potential funders and other agencies to evaluate the merits of the project based on anticipated outcomes and costs.

Planning should include a horizon of at least nine years, covering phases that include establishment, capacity building, implementation and biosecurity. Field activities should include the establishment of pilot or demonstration projects in which personnel can cooperate in research and trials, undertake training and refine management techniques and procedures.

The eradication phase should be organised in steps, clearing areas progressively zone by zone, and initiating active surveillance, to either confirm eradication or improve the process.

Eradicating invasive beavers from southern Patagonia presents special challenges associated, in particular, with the involvement of two countries, the presence of beavers in both continental and insular habitats, and the remoteness and size of the management area. Key issues associated with these challenges include the need to develop efficient and effective governance that reflects the necessary political support for making and implementing decisions and for securing and allocating funds. The development of an effective, goal-oriented management structure that can respond to logistic challenges imposed by these risks will be essential.

Beaver eradication in southern Patagonia is a novel and ambitious project. It will require the development and application of innovative tools and approaches. At the same time, it will allow Chile and Argentina, together with international players, to develop a new cooperative model to handle complex environmental problems. If effective in Southern Patagonia, similar collaborative models may help to improve the management of other global threats to biodiversity.

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Strategy to control the invasive alien tree *Miconia calvescens* in Pacific islands: eradication, containment or something else?

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Abstract: *Miconia calvescens* (Melastomataceae) is a notorious plant invader in the tropical islands of French Polynesia, Hawaii and New Caledonia. A small tree native to Central and South America, it was first introduced as an ornamental in private botanic gardens in Tahiti (1937), Honolulu (1961), and Nouméa (1970s) where it escaped, became naturalised, and formed dense monospecific stands. More than 80,000 ha are currently invaded in French Polynesia, 10,000 ha in the Hawaiian Islands and 140 ha in New Caledonia. Control programmes have been under way in the Hawaiian Islands (Oahu, Maui, Hawaii, Kauai) and French Polynesia (Raiatea, Tahaa, Nuku Hiva, Fatu Hiva) since the early 1990s, and in New Caledonia (Province Sud) since 2006. Despite more than 15 years of intensive control efforts and millions of plants destroyed, eradication has not been achieved in any of these islands, mainly because the species has multiple features that thwart its elimination (e.g., prolific seed production, active dispersal by alien and native frugivorous birds, large and persistent soil seed bank, shade-tolerance), combined with the difficulty of detecting and destroying plants on rough terrain and steep slopes, insufficient control frequency, and limited financial and human resources. Miconia's life cycle requires at least four years growth from seedling to fruiting. Consequently, prevention of fruit production may be an effective management strategy for small populations. This "juvenilization" process may allow the eradication of small populations when carefully conducted over a quarter century.

Keywords: management strategy, invasive plant, juvenilization, seed bank

INTRODUCTION

Pacific islands, along with most other islands worldwide, are vulnerable to the establishment and invasion of alien plant species. In some tropical oceanic islands, such as Hawaii and French Polynesia, the number of plant species that have established, formed sustainable populations and reproduce without human intervention (i.e. naturalised; see e.g., Richardson *et al.* 2000) approaches or exceeds the size of the native flora. Some naturalised species have become invasive and alter native ecosystems, cause severe economic losses, or are responsible for the two combined. Furthermore, the rate of species introductions is increasing, enhancing the risk of new invasions. Major cities, such as Honolulu in Hawaii, Papeete in French Polynesia and Nouméa in New Caledonia have international airports and harbours that act as "transport hubs" for people, goods, and plant and animal species, accidentally or intentionally introduced from Asia, Australia, and the Americas. The high human population density and *per capita* gross domestic product (GDP) of Hawaii, New Caledonia and French Polynesia (Denslow *et al.* 2009; Kueffer *et al.* 2010) may partially explain the high proportions of naturalised and invasive alien plants found in these islands. Moreover, these French and US territories support many public and private botanical gardens that were established in the last century to acclimatise "useful" plants from other tropical and temperate countries, including many forestry and ornamental species, some of which are now considered aggressive plant invaders. Management of current and potentially invasive alien plants has now become a priority for Pacific island countries (Sherley 2000; Meyer 2004).

The most cost-effective strategy for managing invasive species is preventing entry into a potential new range (e.g., Wittenberg and Cock 2001). Weed risk assessment, quarantine regulations, and other biosecurity and phytosanitary measures form a first barrier to plant invasion. When a species is already established and naturalised, three management strategies may be appropriate (Carter 2000; Grice 2009): 1) eradication for recently established species or species with a limited distribution; 2) containment for species which are beyond eradication (or where eradication has been rejected as a goal) but still in an early stage of invasion and expanding their range; and 3) control for large and extensive populations ("sustained control" *sensu* Parkes and Panetta 2009; or "maintenance control" *sensu* Hulme 2006) that may include biological control. An alternative option is to do nothing.

Eradication is the "removal of all individuals of a species from an area to which reintroduction will not occur" (Myers and Bazely 2003) or the "permanent removal of discrete populations" (Parkes and Panetta 2009). Eradication is a function of the area over which the weed is distributed and must be searched for repeatedly following control (gross infestation area), and constraints such as site accessibility, plant detectability, the species' characteristics, control efficacy, and funding support (Panetta and Timmins 2004; Parkes and Panetta 2009). Containment and control are sometimes combined because their common aim is reduction of the density of the target species or its rate of spread.

Whether plant eradications are successful depends on the life history traits of the species, including growth rate, reproductive capacities, and dispersal abilities (distance and speed). A major obstacle for plant eradication is the existence of a soil seed bank, which can persist for several years or more.

To demonstrate the importance of plant life history characteristics to an eradication attempt, we report here on the history of miconia (*Miconia calvescens* DC.: Melastomataceae), which is one of the most damaging plant invaders in native forest of Pacific islands. Miconia is a small tree unlike the "agricultural weeds" such as grasses, herbs, vines, shrubs, and aquatic plants, which are targeted for eradication in California, USA (Rejmanek and Pitcairn 2000), Australia (Woldendorp and Bomford 2004; Parkes and Panetta 2009), or New Zealand (Harris and Timmins 2009). The species is capable of prolific reproduction, has a persistent seed bank (Fig. 1), and can invade species-rich, intact rainforest and cloud forests subsequently destroying native biodiversity. We review the current status and distribution of miconia, and compile the results of control efforts during the past decades in French Polynesia, Hawaii

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and New Caledonia. We discuss the accepted strategies for plant eradication, and propose an alternative strategy to more efficiently manage this species and its environmental threat to all Indo-Pacific tropical high volcanic islands.

CHARACTERISTICS AND INVASION HISTORY

Miconia grows to 4-12 m but may reach 16-18 m in its native range in tropical Central and South America. The species was introduced as a garden ornamental in several private and public botanic gardens worldwide because of its striking, large leaves with purple undersides. It was first introduced to Tahiti in 1937, to the Hawaiian Islands in the early 1960s and to Nouméa in the early 1970s. Historical evidence and molecular analysis (Le Roux *et al.* 2008) indicates that the first plants cultivated in Hawaii and New Caledonia were imported from Tahiti. In each of these island groups, where mean annual rainfall exceeds 2000 mm, miconia escaped from gardens and became naturalised in surrounding vegetation. The lag between introduction and clear signs of invasion in these three island groups has ranged from 20 to 30 years (Meyer 1998), a relatively long time span which may explain why control responses were often too late. The rainforests and cloud forests of all high volcanic islands of French Polynesia and Hawaii, which have relatively similar origins, ages, latitudes, climate, topography and biota, are likely to be under high risk of invasion by miconia. Although New Caledonia is a large continental island with a more subtropical climate and a large area covered by nutrient-poor ultramafic soils, a predictive model shows that miconia might invade up to 25% of Grande Terre rainforests (i.e. 4000 km^2) on sedimentary soils, mainly on the rainy east coast of Province Nord (Meyer *et al.* 2006).

Miconia is already considered to be the most disruptive invasive alien plant in French Polynesia and the Hawaiian Islands, and threatens native rainforests of New Caledonia, the Wet Tropics region of Queensland in Australia (Csurhes 2008), Sri Lanka (Meyer 1998) and some Caribbean islands (Meyer 2010). On Tahiti, Moorea, Raiatea (French Polynesia), Maui and Hawaii (Hawaiian islands) and in Province Sud of New Caledonia, miconia can form dense monospecific stands that suppress native vegetation. Because of its devastating impact on the endemic flora in Tahiti (Meyer and Florence 1996), miconia is viewed as one of the highest control priorities in Hawaii, New Caledonia, and Australia. Potential environmental impacts such as increased runoff and soil erosion, as well as reduced groundwater recharge (Kaiser 2006), make miconia a "transformer species" *sensu* Richardson *et al.* (2000). Miconia was legally declared a "noxious weed" in Hawaii in 1992; a "threat to the biodiversity" in French Polynesia in 1997; a "Class 1 weed", the highest priority category in Queensland, Australia, in 2002; and listed an "invasive exotic species" to be eradicated, by authority of the Code de l'Environnement of the Province Sud, New Caledonia, in 2009.

Ground surveys and helicopter reconnaissance using GPS and GIS have been used to map miconia distribution. Control methods consist of manually uprooting seedlings and saplings, chemically treating the reproductive (or mature) trees on cut-stumps or bark, and carefully targeting spraying from helicopter (the latter only in Hawaii). Volunteers for short-term control operations or long-term funded teams, or both, have been involved and public awareness campaigns have been conducted in all island groups (Conant *et al.* 1997; Medeiros *et al.* 1997; Meyer and Malet 1997; Meyer 2010).

RESULTS

More than 80,000 ha of lowland rainforests and montane cloud forests are currently invaded in French Polynesia, ranging from near sea-level to 1400 m elevation; more than 10,000 ha are invaded in the Hawaiian Islands; and 140 ha in New Caledonia (Table 1). Management programmes detailed below were initiated in French Polynesia on the islands of Raiatea, Tahaa, Nuku Hiva, Fatu Hiva; the Hawaiian Islands on Hawaii, Maui, Oahu, and Kauai beginning in the early 1990s; and in New Caledonia (Province Sud) in 2006 (Table 2).

Raiatea (Society Is., French Polynesia)

Miconia was first introduced in the 1950s as a garden ornamental, then as a soil contaminant in the 1980s. About 250 ha were considered invaded in the early 1990s; infested sites ranged in elevation from sea-level up to 300 m elevation (Meyer and Malet 1997). An eradication attempt was started in 1992. Over 18 years, more than 470 ha has been surveyed (3% of the island surface) and 2.2 million plants have been manually removed, including more than 4500 reproductive trees. More than 3,500 people have been involved, including employees of the Departments of Forestry, Agriculture, Environment and Research, the

Table 1 Miconia invasion in the Pacific islands.
All data from the Hawaiian islands according to "Invasive Species Committees" (BIISC, KISC, MISC, OISC)

Island	Year of introduction	Number of invaded sites or valleys	Elevation range (m)	Invaded area (ha)
FRENCH POLYNESIA				
Tahiti	1937	> 100	10-1400	> 80,000
Moorea	1960s	> 20	10-1100	> 3500
Raiatea	1955	> 10	10-1000	> 470
Tahaa	1980s	1	20-200	< 10
Nuku Hiva	1990s	3	400-1100	< 5
Fatu Hiva	1990s	3	500-600	< 1
HAWAII				
Hawaii	early 1960s	> 100	10-820	> 10,000 (> 45,000*)
Maui	early 1970s	> 20	20-870	> 1000 (>15,000*)
Oahu	1961	> 6	10-500	> 700 (>12,000*)
Kauai	mid-1980s	> 2	40-310	> 220 (>1400*)
NEW CALEDONIA				
Province Sud	1970s	1	200-650	> 140

* surveyed areas including buffer zones of 1 km around all known occurrences, to allow for comprehensive surveillance ("gross infestation area" *sensu* Panetta and Timmins 2004).

Table 2 Results of miconia control efforts in Pacific islands.
Control methods: MC = Manual control; CM = Chemical control; BC = Biological control using the fungal pathogen *Colletotrichum gloeosporioides* f.sp. *miconiae*; (Year) = Year when control started.

Island	Degree of invasion	Control strategy	Control methods	Number of plants destroyed (reproductive trees)
FRENCH POLYNESIA				
Tahiti	High	Control in small areas of high ecological values	MC, CM + BC (2000)	Not evaluated
Moorea	High	Control in small areas of high ecological values	MC, CM + BC (2000)	Not evaluated
Raiatea	Medium	Eradication / Containment	MC, CM (1992) + BC (2004)	2,200,000 (> 4,540)
Tahaa	Low	Eradication	MC, CM (1995) + BC (2005)	10,000 (8)
Nuku Hiva	Low	Eradication	MC, CM (1997) + BC (2007)	8000 (14)
Fatu Hiva	Low	Eradication	MC, CM (1997) + BC (2007)	3000 (5)
HAWAII				
Hawaii	High	Containment	MC, CM + BC (1997)	Evaluation not available
Maui	Medium	Eradication / Containment	MC, CM + BC (1997)	Evaluation not available
Oahu	Low	Eradication	MC, CM (1993) + BC (1997)	16,000 (115)
Kauai	Low	Eradication	MC, CM (1993) + BC (1997)	8000 (23)
NEW CALEDONIA				
Province Sud	Low	Eradication	MC, CM (2006)	170,000 (> 180)

French Army, local volunteers, religious groups, employees of the island Counties, and schoolchildren (Meyer 2010). Campaigns against miconia were organised only once a year because of financial and logistic constraints. The discovery in 2002 and 2003 of isolated, but nonetheless dense miconia populations and reproductive trees at high elevation (up to 1000 m elevation) and in remote gulches and on inaccessible steep slopes, has subsequently shifted the goal to containment.

Tahaa (Society Is., French Polynesia)

A small miconia population was discovered in 1995 in the bottom of a wet valley between 20 and 200 m elevation, near an old track (Meyer and Malet 1997). Reproductive trees and thousands of seedlings have been removed. It is surprising that miconia has not been discovered elsewhere in Tahaa, including the nearby valleys, but detection in dense native *Hibiscus tiliaceus* lowland rainforest is particularly difficult.

Nuku Hiva (Marquesas Is., French Polynesia)

Miconia seedlings were discovered on Nuku Hiva in 1997 during a botanical expedition (Meyer 1998). Three small infestations, between 400 and 1,000 m elevation, have been detected; all originated from soil contamination during road construction. Two of the sites were on very steep slopes, enhancing the difficulty of detection and control. Ground-surveys and a helicopter fly-over were conducted in 2006 (J.-Y. Meyer and R. Taputuarai, unpub. data), but a few mature trees escaped detection in a nearby valley until 2008, after which thousands of seedlings were pulled out (F. Benne pers. comm.).

Fatu Hiva (Marquesas Is., French Polynesia)

Two small infestations of miconia were discovered in 1996 and 2002 by local pig hunters at between 500 and 600 m elevation (Meyer 1998). These populations have few reproductive trees but do contain thousands of seedlings in the understorey of dense native rainforest. Given the locations in the upper portion of a wet gulch, the risk is high that seeds may be washed down rivers. A new population was discovered in 2009 and some non-reproductive plants 4-6 m tall have recently been found at lower elevations (R. Taputuarai pers. comm.). The island's rugged topography makes plant detection and treatment particularly difficult.

Hawaii (Hawaiian Is.)

Miconia was introduced to Hawaii in the 1960s (Medeiros *et al.* 1997). Sustained control did not begin until 1995, due to the large size of the infestation. Comprehensive surveillance on Hawaii would currently need to cover > 45,000 ha (Table 1). Given limited resources, the current strategy involves preventing trees from fruiting along the upper-elevation margin of miconia distribution (J. Leialoha pers. comm. 2009).

Maui (Hawaiian Is.)

Control of miconia began in 1991 and was focused on major infestation sites by 1995. Comprehensive helicopter reconnaissance capable of detecting outlier trees was not initiated until about 2002. The current area surveyed for potential fruiting trees is 15,000 ha, allowing for a 1 km buffer zone around known miconia plants. Two "core" areas totalling about 1000 ha still have fruiting trees. The prognosis seems to be a *status quo* with a large but well-contained miconia population. Containment will require aerial and ground surveillance and control, costing about US$1 million per year, until effective biological control can be implemented.

Oahu (Hawaiian Is.)

Miconia was introduced to the first of three botanical gardens on Oahu in 1961 (Medeiros *et al.* 1997). Two of these gardens (Wahiawa and Waimea) have marginal conditions for its growth with mean annual rainfall between 1500 and 1650 mm. Consequently, spread of miconia was limited, which led to the false belief that the species was innocuous. A single plant introduced to Lyon Arboretum, in Manoa Valley (annual rainfall > 3000 mm) in 1964 produced numerous seedlings that were noted and sporadically removed by staff from 1975. When control began in 1993, there were at least two naturalised populations (Medeiros *et al.* 1997). Fruiting trees on Oahu are currently removed upon detection; 115 have been removed since 1993, including four in 2009 (R. Neville and J. Fukishima, "Oahu Invasive Species Committee" (OISC) pers. comm.). Nearly 12,000 ha needs to be surveyed for miconia, but OISC lacks the resources to survey this entire area, which includes extremely steep topography and narrow valleys.

Kauai (Hawaiian Is.)

Miconia was found in forest on Kauai in 1995 (Medeiros *et al.* 1997), having been introduced in about 1985. An eradication/containment effort was initiated soon afterward. The current management goal is eradication, through detection and removal of potential fruiting trees through surveillance in nearly 1,400 ha, by foot or helicopter. The "Kauai Invasive Species Committee" (KISC) had not seen a fruiting tree from December 2004 to November 2009; however, several fruiting trees were detected and destroyed in late 2009 (K. Gunderson, KISC pers. comm.).

Province Sud (New Caledonia)

Miconia was first introduced from plantings in an 800 ha private botanical garden located above the main town of Nouméa during the 1970s (Meyer 1998). The invaded area is currently estimated to be 140 ha at between 200 and 650 m elevation, which consists of a single major infestation along with isolated trees in small gullies with steep slopes. From 2006 to 2009, 16 ha had been surveyed, and more than 165,000 plants destroyed, including at least six mature trees in 2009. A single isolated plant was discovered and destroyed in 2006 in a private garden at Yienghen 450 km north of Nouméa, but no other plants have been detected since.

DISCUSSION

Can miconia be eradicated?

Despite 4-17 years of intense management and the destruction of millions of plants, miconia has not been eradicated from any of the islands of French Polynesia, Hawaii and New Caledonia, even from small infested areas. We are left asking: why? Eradication success depends on: 1) the number and size of infestations, 2) the accessibility of infestations, 3) detectability of the species, 4) the biological characteristics of the species (or its invasiveness), and 5) effectiveness of the control (Panetta and Timmins 2004).

Furthermore, the most cost effective strategy against invasive plants is early intervention and eradication during a "lag phase" when populations remain small and localised (e.g., Hobbs and Humphries 1995; Loope and Stone 1996). Although news of the effects of miconia invasions on forests in Tahiti reached the Hawaiian islands in the late 1970s, responses to Hawaiian infestations were slow (Medeiros *et al.* 1997). In the Hawaiian islands, miconia had already been introduced to a botanical garden on Oahu and private lands on the island of Hawaii in about 1961. By 1980, miconia was obviously spreading near Hilo, Hawaii, but there was no action against these populations by state or federal agencies. Action began on Maui when miconia was discovered 8 km from Haleakala National Park in 1991, perhaps 20 years after it had been introduced. The concern raised on Maui spread to other islands and by 1995 control of miconia was underway on Maui, Oahu, Hawaii and Kauai.

In New Caledonia, miconia was known to be in a private botanical garden in the early 1990s but since the land owner claimed that the species was locally naturalised but not expanding, there was no control until 2006 when local authorities recognised the need for action. In contrast, there was an immediate response by managers when miconia was discovered in the late 1990s in the Marquesas (French Polynesia). Except in the latter example, early opportunities to eradicate the plant were not taken owing to a general lack of understanding of the threat.

In California, about one third of targeted "weed infestations" between 1 and 100 ha, and one quarter between 101 and 1,000 ha were successfully eradicated during 1972-2000 (Rejmánek and Pitcairn 2002), although biological and ecological attributes of the targeted species were not considered in this analysis. Our results in French Polynesia and Hawaii show that it is unlikely that miconia infestations larger than 500 ha ("net area" *sensu* Panetta and Timmins 2004) can be eradicated with current resources. Eradication may even be difficult with smaller infested areas (Table 3).

Site accessibility and plant detectability are key factors for the success of eradication. Miconia is conspicuous because of its large, bicoloured leaves, but the shaded understorey of dense native rain and cloud forests in French Polynesia, the Hawaiian Islands and New Caledonia limits easy detection of individual plants. The rough topography in these high volcanic islands adds further constraints to eradication.

Miconia life history characteristics and invasiveness

Whether seeds are transient or persistent is fundamental to successful invasive plant management. Eradications may fail for species with seeds that are long lived, buried, rapidly dispersed and spread by uncontrolled vectors such as birds and wind (Carter 2000).

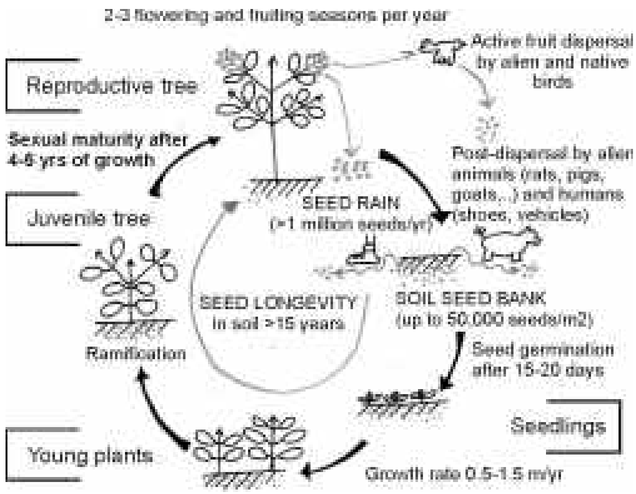

Fig. 1 Miconia life-cycle in Tahiti (*in* Meyer 2010).

Table 3 Proposed miconia control strategy according to the degree of invasion (total infested area and number of infestations).

Degree of invasion	Very localised (1-5 infestations)	Localised (5-50 infestations)	Widespread (> 50 infestations)
Area < 5 ha	Eradication	Eradication / containment?	Containment
Area >5-500 ha	Eradication / containment?	Containment	Containment
Area > 500 ha	Containment	Containment	No control/ biocontrol

Early enthusiasm for eradicating miconia in Hawaii underestimated the persistence its tiny seeds. In the Pacific, a single reproductive miconia produces millions of seeds each year, dispersal is by alien and native frugivorous birds, and a large and persistent soil seed bank is now known to last more than 15 years (Fig. 1). Miconia seeds are only C. 0.5 mm in diameter, so their long seed bank life may be a bit surprising (see Dalling and Brown 2009). The persistence of some invasive species as seeds appears related to the absence of fungal pathogens. For example, fungicide trials with seeds and seedlings of neotropical *Clidemia hirta* (Melastomataceae), which is highly invasive in Hawaii, indicate that fungal pathogens limit growth of *Clidemia hirta* in its native range but not in Hawaii (DeWalt *et al.* (2004). The seed bank longevity of miconia in the Pacific may also result in part from the plant's escape from its native range pathogens. Tropical forest plants, including species of Melastomataceae, are commonly classified into regeneration guilds or functional groups based on their light requirements for seed germination, seedling establishment or growth (Ellison *et al.* 1993). In its invaded range in the Pacific, miconia is a relatively shade-tolerant, late successional, long-lived pioneer, with a large and persistent seed-bank. Its regeneration strategy therefore differs from that of many other invasive trees such as the strawberry guava, *Psidium cattleianum* (Myrtaceae), seeds of which do not live beyond three months in the soil (Uowolo and Denslow 2008).

"Juvenilization", a strategy to control miconia

Control and removal of small populations within a limited area is more likely to be successful than removal over large areas. Moody and Mack (1988) suggest that containment programmes should give priority to small isolated populations ("nascent foci") rather than large infestations. In the case of miconia, small infestations are characterised by many seedlings and few reproductive trees, and large infestations by many reproductive trees and relatively few seedlings and saplings. Since seed production and dispersal rates are high, the management priority is to eliminate all mature trees in all major and minor foci (Fig. 2).

Miconia's "Achilles heel" lies in the four or more years required for growth from seedling to fruiting (Fig. 1). Prevention of the spread of fruit may therefore be an effective strategy for populations small enough to be managed over a long-term with limited resources. This "juvenilization" process is an essential step towards eradication of small populations if maintained for long enough, i.e. beyond the >15 year soil seed bank persistence. This may still seem a long period, but compared with pest animals, the eradication of weeds with long-lived seed populations will often require longer periods of funding and institutional support (Panetta and Lawes 2005). One of the most consistent contributors to success has been gaining widespread, sustained public acceptance of the need for the eradication (Mack and Foster 2009).

CONCLUSIONS

An integrated management strategy incorporating biological control may be the only achievable/sustainable option when miconia populations become so large that eradication is no longer possible; but again, long-term and adequate funding, political will, and institutional commitment are required. Fortunately, effective public awareness campaigns and reinforced biosecurity have prevented the spread of miconia to the other islands with suitable habitat including two high Hawaiian islands (Molokai and Lanai), other Society Islands (Bora Bora, Huahine), Marquesas Islands (Hiva Oa, Tahuata, Ua Huka, Ua Pou), and the southern Austral islands.

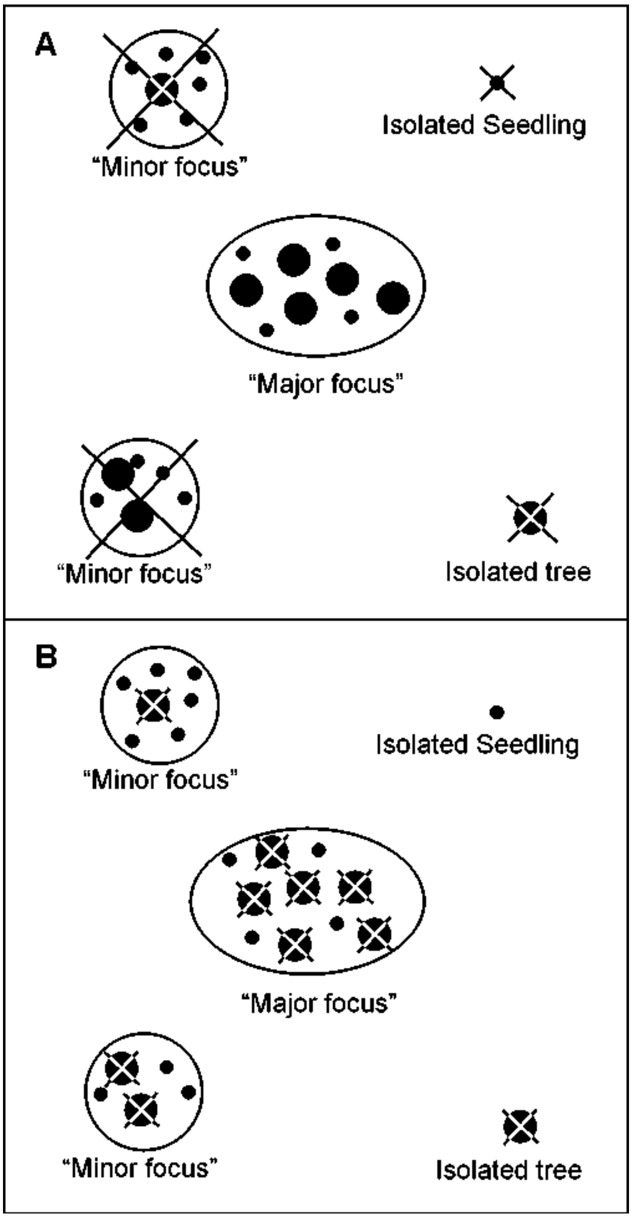

Fig. 2 Common invasive plant strategy following the "Moody & Mack model" (A); "Juvenilization" miconia control strategy (B).

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Non-indigenous freshwater fishes on tropical Pacific islands: a review of eradication efforts

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Abstract Many fishes introduced by humans to islands in the Pacific region have established reproducing populations. As a result, there have been changes to insular aquatic faunas, some obvious and some subtle. Introduced fishes are threats to ecosystems and native species, but little is published about efforts to eradicate them. We compiled information on eradication efforts for freshwater fishes introduced to tropical Pacific islands. Over 60 non-native species of freshwater fishes, representing 18 families, are established on Pacific islands. They represent a diversity of morphologies, environmental tolerances, behaviours, dietary specialisations, and other life-history attributes. We found information on past or present invasive fish eradication efforts for the Hawaiian Islands, Nauru, Kiribati, Palau, Guam, Fiji, and the Galapagos. The main fishes targeted have been tilapias (family Cichlidae) and live-bearers (Poeciliidae). A few eradication efforts succeeded when using chemical ichthyocides, typically rotenone. Future needs include review and modification of existing methodologies to improve efficacy, and development and testing of new chemical and non-chemical methods that may be more selective and less harmful to non-target species.

Keywords: Invasive fishes, Cichlidae, Poeciliidae, Hawaii, Nauru, Kiribati, Palau, Guam, Fiji, Galapagos, tilapia, *Oreochromis mossambicus*, removal methods, physical, chemical, biological

INTRODUCTION

Many species of fishes introduced to islands in the Pacific region have established reproducing populations (Maciolek 1984; Eldredge 2000). Most introductions were associated with aquaculture, commercial and sport fishing, the ornamental fish trade, biological control, and research; some were intentional and others accidental (Maciolek 1984). Introductions of non-native fish in the Pacific began in the 1800s, but newly established species are still being discovered. The introductions have led to marked and often repeated changes to insular aquatic faunas (Jenkins *et al*. 2009), with effects that have often been variable and unanticipated. For instance, the introduction of Mozambique tilapia (*Oreochromis mossambicus*) on many Pacific islands during the mid 1900s was later recognised as disastrous. Among other impacts, it led to the near disappearance of traditional milkfish (*Chanos chanos*) culture (Nelson and Eldredge 1991; Spennemann 2002; Jenkins *et al*. 2009). Moreover, because Mozambique tilapia tolerate high salinity, they also invaded estuaries and other coastal marine environments (Lobel 1980; Maciolek 1984).

Other negative ecological consequences of non-native fishes were illustrated by armoured suckermouth catfishes (family Loricariidae), which are abundant in streams and lakes in Hawaii. The burrows excavated by these species for spawning and nesting destabilise banks and increase erosion (Yamamoto and Tagawa 2000; Nico *et al*. 2009). Other groups such as poeciliids pose multiple threats. These small fishes were initially introduced to the Pacific region for biological control of mosquitoes and later as aquarium releases. Two widely introduced species, the guppy (*Poecilia reticulata*) and western mosquitofish (*Gambusia affinis*), threaten Hawaii's anchialine pool environments (Brock and Kam 1997; Yamamoto and Tagawa 2000). Introduced poeciliids that prey heavily on native aquatic insects likely contributed to the decline or extinction of native stream-breeding damselfly species on Oahu, and the extinction or near-extinction of two other species in Hawaii (Englund 1999). Poeciliids are also the likely source of non-native parasites now present in Hawaiian freshwater ecosystems (Font 2003). Apart from these examples, the ecological and economic impacts of non-native fishes are poorly understood or inadequately documented (Maciolek 1984; Englund 1999). In part, this is because of a lack of field studies (Fuller *et al*. 1999), but even where environmental changes have been observed, cause and effect relationships are difficult to establish.

Because introduced fishes can pose ecological or economic harm (Courtenay and Stauffer 1984; Nelson and Eldredge 1991; Simon and Townsend 2003; Vitule *et al*. 2009), there have been periodic attempts to eradicate some populations (Kolar *et al*. 2010). However, there is little published information about eradication attempts in the Pacific. In part, this reflects the few attempts at removal but there is also evidence that many failed eradication attempts are never published or are otherwise unreported. This is unfortunate because any removal attempts, regardless of the outcome, may provide important insights for future eradication endeavours. Planned eradications that were never attempted may also be useful if they allow other researchers and managers to assess their own current plans, and perhaps reduce the risk of repeating past mistakes. Consequently, more complete knowledge of fish eradication projects in the Pacific region should help improve decision-making processes about how best to use limited resources when dealing with invasive fishes.

In this paper we compile information on past and ongoing plans and projects to eradicate non-native fish populations within the Pacific, largely focusing on smaller islands and island groups near the equator. Much of the information is unpublished. We also briefly describe the diversity of the non-native ichthyofauna as well as the types of inland aquatic habitats invaded along with their native faunas. Such information helps to identify the issues faced when an eradication of invasive fishes is attempted. Lastly, because the methods used in the Pacific to eradicate non-indigenous fishes are only a subset of the methods used elsewhere, we review the global techniques and strategies used to eradicate or control invasive or undesirable fishes.

METHODS

We focused our review on small Pacific islands within the boundaries of the Tropic of Capricorn and the Tropic of Cancer and included obscure literature, agency reports, personal communications, and internet sources. Other information on the diversity of invasive fishes and habitats, details of eradication projects, and methods from other parts of the world were based on an extensive literature review. We supplemented some information from personal experiences over more than 25 years of research on non-native fishes, including some research on fishes in their native ranges. We excluded Pacific islands outside the tropical zone, largely because substantial information about fish eradication and control in places such as New Zealand is readily available in the technical and scientific literature (e.g., New Zealand Department of Conservation 2003; McDowall 2004a; Neilson *et al*. 2004; Nishizawa *et al*. 2006; Yonekura *et al*. 2007).

Positive identification of introduced fishes is often difficult, partly because of unresolved taxonomy and unstable nomenclature of many fish groups. Some taxa, such as the tilapias, are particularly problematic because of frequent hybridisation in captivity and in the wild, as well as the creation of new strains by aquaculture researchers (Costa-Pierce 2003; D'Amato *et al*. 2007). Ichthyologists also periodically re-examine non-native fish specimens and, in some cases, have corrected previous misidentifications (e.g., Courtenay *et al*. 2004). Consequently, some names appearing in past publications are no longer valid.

AQUATIC HABITATS AND NATIVE FAUNAS

Inland aquatic habitats of Pacific islands are diverse, varying dramatically in type, distribution, elevation and coverage (Ellison 2009). Small or low-lying islands typically have few, if any, surface freshwater habitats and therefore are rarely able to support freshwater fish. Larger and more diverse islands rival large continental regions for the diversity of aquatic habitats, many of which are suitable for a wide variety of fish species.

Pacific island drainages are typically small and streams are relatively short compared to continental rivers. Nevertheless, the more topographically diverse islands may contain a wide array of lentic and lotic habitats, ranging from moderately large streams, channels, and ditches to natural and artificial lakes and ponds. Elevated islands often have streams that originate in uplands; cascade down steep slopes and cliffs; contain habitats such as falls, high-velocity runs, rapids, and deep pools; and become estuarine where they empty into the ocean. Waterfalls near the coast can act as barriers, which determine the distribution of some aquatic invertebrates and most fishes (Keith 2003). Temporal differences can also exist. During rainy seasons high-gradient streams become torrential, but during droughts smaller streams are often reduced to a series of isolated pools.

The diversity and abundance of native fishes and aquatic invertebrates varies greatly among the different Pacific island groups (Donaldson and Myers 2002; Keith 2003; McDowall 2004b). Many species are unique (endemic) to particular islands or island groups, with some only in specific habitats (Brock and Bailey-Brock 1998; Keith *et al*. 2002; Keith 2003). Aquatic invertebrate groups native to the Pacific islands can be quite diverse. By comparison, native freshwater fish faunas are generally depauperate. Indeed, some island lakes and streams that are naturally devoid of native fishes support a diverse fauna of invertebrates. Much still remains unknown about the inland aquatic faunas of Pacific islands; field studies continue to yield new information on the natural history and biology of native species as well as the discovery of new species (Keith 2003; Englund 2008).

Many of the native fishes present in streams on Pacific islands belong to families that are predominantly marine. The life-history strategy among most such groups (e.g., sicydiine gobies and eleotrids) is amphidromy, where juveniles feed and adults spawn in freshwater habitats and larvae are carried to estuaries or the sea (Keith 2003; McDowall 2007). In contrast, adults of catadromous species (e.g., anguillid eels) spawn at sea and sub-adults migrate to freshwater habitats. Many native inland fishes and invertebrates of Pacific islands have restricted ranges, small population sizes, and no natural defences against invaders so they are vulnerable to extirpation or extinction where non-native fishes become established.

DIVERSITY OF NONINDIGENOUS FISHES ON PACIFIC ISLANDS

Most non-native freshwater fishes established in the Pacific are found on the larger islands because these sites offer a diversity of aquatic habitats, including many places suitable for invasion. In the Pacific, non-native fishes commonly occur in heavily disturbed sites (e.g., roadside ditches and artificial reservoirs), but some are also found in relatively pristine habitats (e.g., caldron lakes and mountain streams). On large, diverse islands such as Oahu (Hawaii) and Guam, non-native fish abundance in certain habitats, such as some natural streams and artificial reservoirs, are often at densities far exceeding those of native fishes present (Yamamoto and Tagawa 2000; L. G. Nico pers. obs.). Much less vulnerable to invasion are the many small, low-lying Pacific islands, because these areas have few freshwater habitats.

Four publications review information on non-indigenous fishes of the Pacific region. In a comprehensive analysis of introduced freshwater fishes in the Hawaiian Islands and other tropical islands of Oceania (excluding New Guinea and the region south of the Tropic of Capricorn), Maciolek (1984) documented 41 non-marine fish species representing 14 families. A review by Nelson and Eldredge (1991) focused on the widely introduced tilapiine cichlids, and detailed their distribution and status on islands throughout the South Pacific and Micronesia. The information on the status of introduced fishes established in Hawaii (Maciolek 1984) was updated by Devick (1991) and Eldredge (2000) added new data, provided information for New Guinea and identified 86 freshwater fish species introduced into fresh and brackish waters in the region. However, it remained unclear how many species were considered to be established. Our review of the Eldredge checklist (which inadvertently excluded loricariid catfishes) revealed that at least 62 species of freshwater fish representing 18 families have become established in the Pacific islands.

This remarkable range of taxa includes those that originated from Asia, Africa, Europe, and South, Central and North America. The most widely introduced fish families are Cichlidae (e.g., tilapias) and Poeciliidae, each with up to 9 species established. Other families include Centrarchidae (black basses and sunfishes), Cyprinidae (carps and minnows), and Loricariidae (suckermouth armoured catfishes). The most widely introduced species include Mozambique tilapia, one or more species of mosquitofish (*Gambusia*), guppy, and common carp (*Cyprinus carpio*).

Some introduced species are tropical and others from temperate climates. Most primarily inhabit fresh water, but others are euryhaline and able to survive and/or reproduce in fresh, brackish and marine environments. A few are air-breathing fish (e.g., synbranchid eels, loricariid and clariid catfishes) and able to persist in habitats nearly devoid of dissolved oxygen. Body size ranges from the guppy, with adult males typically < 2.5 cm total length, to the Asian carps (e.g., grass carp, *Ctenopharyngodon idella*), which commonly grow to well over one meter. Nearly all major trophic levels are represented, including small and large herbivores, omnivores, and predators. The herbivores include some that specialise on phytoplankton (e.g., silver carp; *Hypophthalmichthys molitrix*), attached algae (e.g., loricariid catfishes), and macrophytes (e.g., grass carp). Among carnivores, some species prey mostly on fishes and other vertebrates (e.g., members of the genera *Cichla* and *Channa*), whereas others, typically smaller predators, normally consume invertebrates, including insects and small crustaceans (e.g., oriental weatherfish, *Misgurnus anguillicaudatus*).

NON-NATIVE FISH ERADICATION PROJECTS IN THE TROPICAL PACIFIC

There are few documented accounts of invasive fish eradication or control projects for the tropical Pacific. A few published articles mention fish control operations for selected Pacific islands, but usually lack details. Here we review information on attempted or planned invasive fish eradications for seven islands or island groups in the tropical Pacific (Table 1): the Hawaiian Islands, Nauru, Kiribati, Palau, Guam, the Galapagos, and Fiji (Fig. 1).

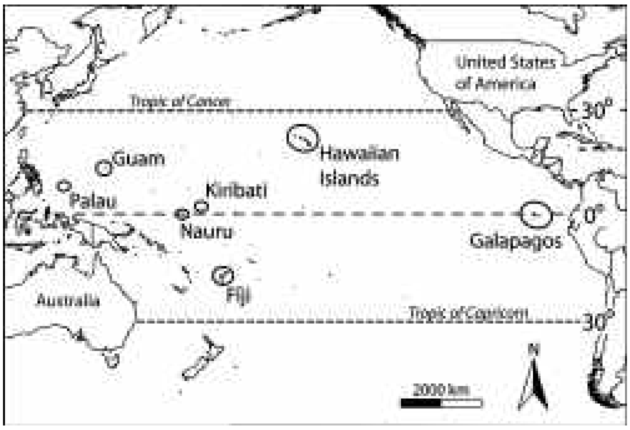

Fig. 1 The Pacific Ocean showing locations of the seven island groups where documented non-native fish eradication or control projects have occurred.

Hawaiian Islands

There has been emphasis on research and assessment of the spread and impacts of invasive aquatic organisms in the Hawaiian Islands (Eldredge 1994; Yamamoto and Tagawa 2000; Englund 1999, 2008). However, not until the past one or two decades has removal been considered regularly as a management option. The literature indicates toxicants had never been used for fisheries management in Hawaii prior to about 1970 (Lennon et al 1971) and, although eradication was discussed (Doty 1974), there were no known fish eradication projects from 1965 to 1979 (J. Maciolek pers. comm.).

The more serious attempts to eradicate invasive fish in Hawaii have focused on anchialine pools, which are small, landlocked water bodies near the coast and only with subterranean connections to the sea (Brock and Kam 1997; Yamamoto and Tagawa 2000; Santos 2006). Such pools are largely associated with geologically young lava fields and therefore they are most abundant on the highly volcanic Big Island of Hawaii (Yamamoto and Tagawa 2000). Anchialine pools are influenced by tides and commonly contain brackish water, although salinities may vary within and among pools depending on their distance from the ocean and amount of freshwater inflow (J. Maciolek pers. comm.). These pools represent unusual ecosystems, in part because they are inhabited by endemic native invertebrates, including some that are imperilled species (Brock and Kam 1997; Yamamoto and Tagawa 2000; Santos 2006). The Hawaiian Islands probably have the greatest number of anchialine pools in the world, but many have been modified or destroyed in the last 60 years due to a combination of coastal development and introduction of non-native species (Brock and Bailey-Brock 1998, Santos 2006).

Over 95% of existing anchialine pools of Hawaii are invaded by non-native fishes, primarily poeciliids and tilapia (Yamamoto and Tagawa 2000). In these pools the poeciliids (mainly western mosquitofish and guppy) negatively impact native shrimps or "opae'ula" (*Halocaridina rubra*), apparently through direct predation, habitat displacement (by driving the shrimp into underground fissures and crevices), or both (Brock and Kam 1997; Yamamoto and Tagawa 2000). Opae'ula shrimp are minute (< 15 mm long) herbivores and in anchialine pools may be a keystone species because of their heavy grazing on attached algae. Declines of opae'ula shrimp in the presence of poeciliids are followed by overgrowth of algae, changes in the dominance of algal species, and declines in native invertebrates (Brock and Kam 1997; Capps *et al*. 2009).

Brock and Yam (1997), without providing precise locality or date information, reported that they successfully used rotenone to remove non-native fishes (presumably poeciliids) from some relatively isolated anchialine pools. More recently, at Kailua-Kona (Fig. 2a) on the island of Hawaii, rotenone was used with similar success, with evidence that the full complement of native species rapidly recovered (Chai and Mokiao-Lee 2008). However attempts elsewhere succeeded against tilapia but failed for mosquitofish possibly due to reinvasion via an underground link to a nearby artificial pond. Rotenone is considered toxic to organisms that respire through gills, but native invertebrates such as opae'ula shrimp often re-colonised treated anchialine pools from their underground refuge even before rotenone fully degraded (Brock and Yam 1997; Yamamoto and Tagawa 2000).

Rotenone was suggested as the most efficient way to remove tilapia and guppies in two anchialine pools on private property near Kiholo Bay (Hawaii Department of Land and Natural Resources 2000), but it is unknown if the removal effort was ever attempted. The use of rotenone is often controversial in Hawaii. Those wanting to use the toxicant in open waters for invasive fish removal typically encounter problems obtaining official permission. For example, the Malama Kai Foundation received funding from the National Oceanic and Atmospheric Administration in 1999 to restore certain anchialine pools on the island of Hawaii. Restoration was to include removal and control of non-native species, but, because the pools had subterranean connections, removal of non-natives by manual methods was not feasible. According to available information, the Foundation was unable to secure state permission to use rotenone, thereby stalling restoration efforts (http://www.malama-kai.org/management/ponds.htm).

Upland streams in Kokee State Park, Kauai Island, have been invaded by rainbow trout (*Oncorhynchus mykiss*), but restoring these streams to their natural fishless condition would necessitate use of a chemical ichthyocide (Englund and Polhemus 2001). However, public acceptance for such a project seems unlikely because the use of poison, particularly rotenone, would likely harm non-target indigenous and endemic aquatic arthropods (Englund and Polhemus 2001). Furthermore, Englund (2008) concluded that the use of ichthyocides would likely be unsuccessful where invasive fishes present (i.e., poeciliids and tilapia) can survive in high-salinity coastal waters and ultimately re-invade streams following chemical treatment. The use of toxicants at sites such as Kane'ohe Bay would also encounter technical problems because of the large size of the bay, and public resistance. However, eradication might be possible in high-gradient streams that terminate into the ocean via high waterfalls, because the falls would function as barriers preventing re-invasion by any non-native fishes escaping to coastal waters (Englund 2008).

As an alternative to chemicals, a state biologist in Hawaii investigated the possibility of importing male pike killifish (*Belonesox belizanus*), a small piscivorous poeciliid fish native to Central America, with the intent of releasing a few into anchialine pools as a biological control against other, but smaller, non-native poeciliids (M. Yamamoto pers. comm.). It was believed that pike killifish, a surface dweller, would preferentially prey on other poeciliid fishes and generally avoid bottom or cave

Table 1 Non-native fish eradication and control attempts in the tropical Pacific.

Group	Targeted taxa	Habitat and site	Method (Year)	Outcome	References
Hawaii	Poeciliid fishes	Anchialine pools	Rotenone (1990s?)	Success	Brock and Kam (1997)
	Western mosquitofish	Anchialine pools; Kailua-Kona (Hawaii)	Hand nets, seines, traps Rotenone (2007)	Failed Success	Chai and Mokiao-Lee 2008; Carey *et al*. 2011; D. Chai (pers. comm.)
	Western mosquitofish plus tilapia	Anchialine pool; Wai'olu (Hawaii)	Rotenone 5 ppm (2008) and later at higher concentration	Success on tilapia; failed on mosquitofish	Carey *et al*. 2011; D. Chai (pers. comm.)
	Loricariid catfish *Pterygoplichthys*	Waihawa Reservoir (Oahu)	Back pack and boat mounted electroshockers	Failed	M. Yamamoto (pers. comm.)
Nauru	Mozambique tilapia	Inland ponds and brackish lagoons	Rotenone	Mixed success	Ranoemihardjo 1981; B. Ponia (pers comm.)
Kiribati	Mozambique tilapia	Temaiku fish farm	Rotenone, seine nets; increased fertility using fertiliser and decaying tilapia (1982)	Failed	Teroroko 1982, 1990
Palau	Mozambique tilapia	Four ponds on Malakal, three fresh, one brackish	Non chemical methods including explosives (2003) Levels reduced by pumping, then rotenone and perhaps some chlorine (2004)	Failed Succeeded at three sites	EQPB 2004, GISD 2006; E. Edesomel (pers. comm.)
Guam	Hybrid tilapia presumably *Oreochromis mossambicus* x *O. urolepis hornorum*	Small reservoir	Illegal poisoning, chemical unknown	Success (?)	Maciolek 1984
	Chevron snakehead	River catchment	Physical methods including baited drop lines, seine nets and dip nets	Incomplete	B. Tibbatts (pers. comm.)
Galapagos	Nile tilapia	Freshwater crater lake	Rotenone (2008)	Success	L.G. Nico (unpublished data)
Fiji	Juvenile Mozambique tilapia	Two ponds filled with seawater	Biological control using a predator, Hawaiian ladyfish (early 1970s or before)	Partial success	Popper and Lichatowich 1975

areas where native shrimp normally occur. Moreover, it was reasoned that an all-male pike killifish population, unable to reproduce, would naturally die off within a short period. Given that mosquitofish and other established poeciliids were already preying on native shrimp and other invertebrates, supporters of the plan argued that the introduction of a few non-reproducing predatory fish was worth the risk. However, proponents of the plan were unable to convince the Hawaii Department of Agriculture to change the legal status of pike killifish from its existing designation as a prohibited species to a less restricted status that would allow its import for research purposes.

There has not been much contemplation of fish eradication in Hawaii outside anchialine pools even though non-native fishes are abundant in many of Hawaii's lakes and streams including suckermouth armoured catfishes (family Loricariidae) in the genera *Pterygoplichthys*, *Hypostomus* and *Ancistrus* (Yamamoto and Tagawa 2000). Electroshockers have proved ineffective against these catfish, presumably because the electrical field does not penetrate into their burrows (Table 1).

According to R. Englund (pers. comm.), dewatering is performed regularly by the U.S. Fish and Wildlife Service in Hawaii at Hanalei National Wildlife Refuge (Kauai Island) and Pearl Harbor National Wildlife Refuge (Oahu Island) to rid taro fields of tilapia. The U.S. National Park Service is planning to explore alternatives on how best to eradicate tilapia from historical fish ponds on the island of Hawaii.

Nauru

Mozambique tilapia were introduced to Nauru circa 1960 for mosquito control and as a food fish (Ranoemihardjo 1981; Fortes 2005). The species rapidly expanded its range throughout the island, competed with native milkfish for food and space, preyed on young milkfish, and caused a decline in Nauru's traditional milkfish culture (Ranoemihardjo 1981; Nelson and Eldredge 1991; Spennemann 2002). At the request of the Republic of Nauru, the Food and Agriculture Organization (FAO) of the United Nations initiated a tilapia eradication program in 1979-1980. Methods considered included complete drying of selected smaller ponds, stocking predatory fish as a biological control, removal of tilapia with nets and traps, and application of fish toxicants (Ranoemihardjo 1981). Following bioassay tests to determine adequate concentration, rotenone was applied to a series of ponds and lagoons, with mixed success. Although Ranoemihardjo (1981) concluded that repeated rotenone application would eventually eliminate remaining tilapia, there were problems resulting from a shortage of manpower and equipment, and the onset of the rainy season. Later authors described the 1979-1980 eradication effort as unsuccessful (Nelson and Eldredge 1991; Thaman and Hassall 1996; Fortes 2005).

Mozambique tilapia remain a problem in Nauru because they commonly re-invade previously treated ponds. A practical strategy for dealing with the species may require a national tilapia plan that includes policies, education and training, polyculture, and other potential species for use in aquaculture (Fortes 2005). Some ponds cleared of Mozambique tilapia were later stocked with Nile tilapia (*Oreochromis niloticus*), which are considered more desirable as a culture fish by many aquaculture proponents. The Secretariat of the South Pacific Community (SPC) believes that complete eradication of Mozambique tilapia in Nauru and other small Pacific islands would be difficult or impossible, and probably not worth the resources required. The alternative is population control aimed at: 1) preventing spread of Mozambique tilapia and other non-native fishes; 2) removing Mozambique tilapia from ponds or aquaculture areas where it is considered a nuisance and competitor; and 3) identifying and protecting areas known to contain endemic or otherwise threatened local populations of indigenous species (B. Ponia pers. comm.). The SPC is also considering introducing Nile tilapia into areas occupied by Mozambique tilapia, hoping that the two species will hybridise into a more desirable aquaculture food fish.

Kiribati

As for other Pacific islands, the introduction of Mozambique tilapia to Kiribati has negatively affected the culture of milkfish (Gillett 1989). However, attempts to eradicate tilapia have been unsuccessful (Teroroko 1982; Eldredge 1994). According to Maciolek (1984), the Republic of Kiribati considered that tilapia required major eradication effort by its Department of Natural Resources. We found no recent updates of this situation, whether the eradication efforts (Table 1) described by Teroroko (1982) continue or whether the tilapia population on the island has declined. In a 2002 fishery country profile for Kiribati, the FAO reported that an 80-ha milkfish farm established on South Tarawa in the late 1970s was unproductive, partly because ponds contained introduced tilapia (FAO 2002).

Palau

In 2003, tilapia were found in water bodies on Palau's island of Malakal and identified by one of us (LGN) as Mozambique tilapia, although introgressive hybridisation could not be ruled out (specimens catalogued as UF 163824, ichthyological collection, Florida Museum of Natural History). In December 2003, the President of Palau declared a "Quarantine Emergency" in response to which Palau's Bureau of Agriculture coordinated the use of ichthyocides. These were applied in early 2004 by a multi-agency team led by staff of the Palau Environmental Quality Protection Board (EQPB) at the four sites containing tilapia.

Three of the sites were fresh water, each covered 0.1 to 0.2 ha and included two in close proximity known as the "Japanese fuel tank" or "barrack" ponds (Fig. 2b) and the third in a rock quarry site on Palau Transportation Company property (Fig. 2c). The rock quarry site was characterised as a complex of small water bodies, including a quarry pond, two smaller retention ponds, a puddle and an overflow stream. The last of the four sites was a large rectangular (150-m x 25-m; 0.4 ha), brackish-water pond along the northeast coast of Malakal Island constructed as a dry dock by the Japanese during World War II (Fig. 2d). In 2004, the four sites were treated with rotenone (supposedly in conjunction with chlorine at one site) resulting in recovery of at least 38,800 dead tilapia (EQPB 2004), although many more dead were not recovered (E. Edesomel pers. comm.).

In January 2006, the Quarantine Emergency was lifted and the government declared that "no known infestations" of tilapia existed in the country (EQPB 2004; GISD 2006). However, new reports of tilapia in the rock quarry pond in 2006 were verified by EQPB. It was uncertain whether

Fig. 2 Inland water bodies on Pacific islands treated with chemicals for purpose of eradicating invasive fish: anchialine pool (A) on Big Island of Hawaii where rotenone was used to remove non-native poeciliids; three artificial ponds (B-D) in Palau where rotenone or chlorine was used to remove Mozambique Tilapia. All attempts were successful, except for site C, the quarry pond (see text for additional information). Photographs by David Chai (A); William Barichivich (B and D), and L. G. Nico (C).

that discovery represented a new, separate introduction or was of fish that survived the 2004 rotenone treatment (GISD 2006; PNISC 2006). Rotenone reapplied to the quarry pond during 2006-2007, killed at least 300 additional tilapia, mostly small juveniles (PNISC 2006; PIICT 2009). During our visit to the quarry pond in early 2010, we captured and preserved a few juvenile specimens, an indication of continued tilapia reproduction.

Guam

Maciolek (1984:147) reported that hybrid tilapia (presumably *Oreochromis mossambicus* x *O. urolepis hornorum*) were stocked into a small reservoir on Guam, but noted the fish were later eliminated as a result of "illegal poisoning." Details are lacking, so it remains unclear the type of chemical involved. Guam Division of Aquatic and Wildlife Resources (DAWR) personnel are currently attempting to remove introduced chevron snakeheads (*Channa striata*) from the Ajayan River drainage in southern Guam; a population present since the 1970s as a result of escapes from a local aquaculture facility (B. Tibbatts pers. comm.). DAWR biologists do not use fish toxicants and are reluctant to use electrofishing gear because of concerns of harming native eleotrids and gobiid fishes.

Galapagos

In 2006, a reproducing population of Nile tilapia was discovered in a natural freshwater crater lake in the Galapagos Archipelago of Ecuador (L. G. Nico unpubl. data). Galapagos National Park authorities decided on use of rotenone and U.S. Geological Survey biologists were asked to assist in the eradication. In early 2008 rotenone was applied and approximately 40,000 dead and dying tilapia were removed from the lake. Prior to application of rotenone, aquatic invertebrates were collected and held in nearby refuge tanks. After removal of the tilapia, and once all residual rotenone in the lake had degraded sufficiently, captive invertebrates were released back into the lake to speed recovery of invertebrate communities that might have been affected by the chemical. The eradication was considered a success and a paper describing the project in detail is in preparation.

Fiji

During the early 1970s, perhaps before, experiments were conducted in two seawater ponds on Fiji using the predatory Hawaiian ladyfish (*Elops hawaiensis*) to control small juvenile Mozambique tilapia (Popper and Lichatowich 1975). After about 70 days, it was concluded that no tilapia fry were present in the small (0.2 ha) pond and that juvenile tilapia numbers were reduced in the larger (2 ha) pond, but we found no information to indicate the methods were ever applied on a broader scale or for eradication. Although details are scant, the study was apparently conducted with the aim of reducing interspecific competition of tilapia so as to improve their culture, rather than for the purpose of eradicating the non-native fish.

FISH ERADICATION METHODS: STATE OF KNOWLEDGE

Throughout the world, attempts to eradicate non-native fish populations have had widely mixed results (Cailteux *et al*. 2001; Kolar *et al.* 2010). No single known eradication method succeeds in all environments or for all fish species, although much new knowledge has been gained over the past few decades. Most successful eradications relied on fish toxicants, mainly rotenone (Britton *et al*. 2009). However, the use of these ichthyocides has often failed, although some failures were likely the result of poor planning or inadequate implementation. Rinne and Turner (1991) evaluated 26 projects that used toxicants to remove unwanted fishes from streams in the western United States (USA). Nine (35%) projects were judged to be "successful," 15 (58%), were "unsuccessful" or "failures," and two were "short term success" or of "variable success." Meronek *et al*. (1996) assessed 51 projects that used physical and/or chemical methods control one or more target fish species, and judged 32 to be successful. However, their definition of success did not necessarily mean eradication.

Globally, few entire populations of invasive species of fish have been targeted for eradication and, among those, few were successful. The few successes have been in small, shallow, easily accessible, sparsely vegetated, closed aquatic systems such as ponds or small lakes. Eradication in more open or complex systems such as large streams and wetland habitats is generally impossible or, at best, difficult and expensive. Whether eradication of non-native fishes is a viable option, the degree of difficulty depends on factors such as the type, abundance, and geographic distribution of the targeted species plus the physical and biological composition, size, complexity, and sensitivity of the invaded environment (Kolar *et al*. 2010). Also to be considered are: the existence of, or potential for, development of reliable methods, and availability of funding, human power, expert leaders and trained crews (Donlan and Wilcox 2007). Appropriate planning requires clear identification of goals or criteria to be met before eradication proceeds (Chadderton 2003). This may involve implementation of an adaptive management strategy (Gehrke 2003; Kolar *et al*. 2010).

Successful eradication requires some basic knowledge of the targeted species and the invaded environment. A critical first step is positive species identification in part to confirm that it is truly non-native (Fuller *et al*. 1999). Following confirmation of an invasion, rapid but comprehensive field surveys using appropriate gear are needed to ascertain its geographic extent. If eradication is deemed viable, it is essential to rapidly gather basic information on abundance, reproductive status and strategies, life history, environmental tolerances, and population dynamics. Such information may provide clues about a non-native species' characteristics that may be targeted or otherwise useful for developing the eradication effort.

In general, methods for eradication of invasive fishes can be divided into three categories: chemical, physical, or biological. An integrated approach is often chosen, using multiple methods in combination (Lee 2001; Diggle *et al*. 2004; Kolar *et al.* 2010). Many invasive fishes have high reproductive potential and the survival of even one adult pair can potentially lead to thousands of offspring. For this reason, spawning grounds are often a primary target of both eradication and control efforts (Diggle *et al*. 2004).

Chemical methods

Fish toxicants (i.e., ichthyocides, piscicides, or fish poisons) are the primary method for eradicating invasive fishes, with more than 40 different chemicals used worldwide (Kolar *et al.* 2010). Most such products have not been fully developed or tested, many are not approved for fish management and only a few are widely and consistently used (Dawson 2003; Clearwater *et al*. 2008; Cailteux *et al*. 2001; Kolar et al. 2010). The most commonly used ichthyocides are rotenone and Antimycin-A (Fintrol®). We have not compiled information on the legal status of rotenone and other fish toxicants for the many Pacific island governments. However, guidelines for the effective and safe planning and execution of projects using rotenone are widely available (Finlayson *et al*. 2000; Moore *et al*. 2008). The American Fisheries Society also has a Rotenone Stewardship Program and periodically offers training courses on how to plan and execute rotenone and antimycin projects (see http://www.fisheries.org/units/rotenone/).

Rotenone is naturally found in plants of the family Leguminosae and is the active ingredient in some plants used by early Pacific islanders as a poison in the harvest of food fish (Morrison *et al*. 1994). In North America, rotenone has been used by fish biologists as a piscicide since the 1930s against numerous fish species and in habitats ranging from still to flowing waters (Rinne and Turner 1991; McClay 2005). There is now a substantial literature on the use of this toxicant (Wydoski and Wiley 1999; Cailteux *et al*. 2001; McClay 2005).

Antimycin is a fungal antibiotic recognised for its potential use in fish management since the early 1960s (Finlayson *et al*. 2002; Moore *et al*. 2008). Rotenone and antimycin are both general piscicides, but depending on the habitat and fish species to be controlled, they have sometimes been used selectively (Willis and Ling 2000; Moore *et al*. 2008). For example, scaled fish and some rotenone-resistant species are often susceptible to antimycin (Finlayson *et al*. 2002). Because efficacy depends on water and habitat characteristics (e.g., pH, water flow, and amount of leaf litter), antimycin is sometimes used in small streams whereas rotenone is used in large, deep lakes (Finlayson *et al*. 2002). Application of each chemical typically involves release of diluted liquid solutions directly into the water, although rotenone powder is commonly used. There has also been research on ingestible, feed pellets (poison bait) containing rotenone or antimycin, (Mallison *et al*. 1995; Kroon *et al*. 2005).

The main advantages for antimycin are its effectiveness at lower concentrations and non-detectability by fish, whereas rotenone has the advantages of broad range of toxicity to all species of fish and effectiveness under a wide range of pH conditions (Finlayson *et al*. 2002). Rotenone is generally much less expensive than antimycin. Both chemicals degrade relatively quickly into harmless compounds and are neutralised by potassium permanganate (Moore *et al*. 2008). Depending on water temperature and sunlight exposure, degradation may be within days or weeks for rotenone or within hours or days for antimycin (Dinger and Marks 2007). Depending on concentration, both chemicals can be harmful to aquatic invertebrates, especially those that have gills. However, much less is known about the non-target effects of antimycin (Finlayson *et al*. 2002; Dinger and Marks 2007).

Less studied, potentially useful toxicants include a diverse group of plant-derived saponins or triterpene glycosides, including certain products listed in the literature as teaseed cake and Mahua oilcake (Clearwater *et al*. 2008). Additional promising ichthyocides include squoxin, selective against northern pikeminnow (*Ptychocheilus oregonensis*) and several others because of their apparent selectivity, low toxicity to non-target organisms, ease of application, safety to humans, persistence in the environment, low tendency to bioaccumulate, and low cost (Dawson 2003). However, although there is a need and continued interest in developing these and other new piscicides, costs and time associated with research and registration may preclude their availability in the near future.

Most fish toxicants have the disadvantage of non-specificity, causing death or harm not only to targeted non-native fish but also non-targeted native fishes and aquatic invertebrates. Many non-native species are less sensitive to ichthyocides than the non-target species (Schofield and Nico 2007; Schreier *et al*. 2008). Fish toxicants that kill native species, commonly require restocking to offset their effects, although in some tropical insular Pacific habitats this is often unnecessary because native fishes and macroinvertebrates reinvade naturally from coastal areas or nearby inland drainages. However, caution is necessary especially since simultaneous chemical treatment of more than a few streams could eliminate non-migratory stream invertebrates from the entire island. Furthermore, the unwise use of fish toxicants in drainages containing imperilled native species may have disastrous results (see Holden 1991).

Physical methods

Nets, traps, gigs, spears, electrofishing gear, explosives, and management of water levels and flows are all physical methods used to control invasive fish populations. Most of these have limited potential for eradication (Roberts and Tilzey 1996; Wydoski and Wiley 1999; Mueller 2005; CDFG 2007; Kolar *et al.* 2010).

Eradications using nets and traps are limited to small, isolated water bodies or portions of drainages. For instance, intensive seining during 1976-1978 reportedly removed all non-native sheepshead minnow (*Cyprinodon variegatus*) and its hybrids from a small stream system in Texas, USA (Minckley and Deacon 1991). Gill netting helped to eradicate non-native trout from high mountain lakes in California, USA (Knapp and Matthews 1998; Vredenburg 2004) and Banff National Park in Canada (Parker *et al.* 2001). Small traps were used to eradicate non-native fish from an isolated pool in Mexico (Lozano-Vilano *et al.* 2006). In contrast, tests of gill nets in New Zealand ponds (Neilson *et al*. 2004) failed to eradicate or control rudd (*Scardinius erythrophthalmus*).

Backpack electrofishing gear has been tested for removal of non-native salmonid populations in small upland streams in North America with mixed results (Moore *et al.* 1986; Thompson and Rahel 1996; Kulp and Moore 2000). Electrofishing (by boat or backpack) may be useful for control but not eradication in larger or more complex water bodies. For example, boat-mounted electrofishing gear has been deployed regularly since 2001 to remove Asian swamp eels (Synbranchidae) from canals in south Florida, USA. Approximately 1,400 swamp eels were removed the first year but results appeared to have little initial effect on overall population size or size-length structure (L. G. Nico, unpubl. data).

Underwater explosives such as detonation cord can kill or injure fishes (Teleki and Chamberlain 1978; Keevin 1998), but is expensive and largely ineffective for eradication of invasive species (CDFG 2007). Considerable variation exists in blast effects depending on charge type (e.g., low-velocity versus high-velocity detonation; linear versus point source), charge weight, blast design (e.g., detonation depth), and habitat characteristics (e.g., depth and bottom configuration) (Keevin 1998). Vulnerability to explosives (i.e., mortality rate and severity of injury) also varies between fish species. Fish with gas bladders (buoyancy organs) suffer great harm whereas those that lack gas bladders (e.g., swamp eels) often survive underwater explosions (Goertner *et al.* 1994).

Fishes can exhibit differences in thermal tolerance, but manipulation of water temperature to eradicate or control non-native fish is seldom feasible. In a rare example, Stauffer *et al.* (1988) determined that the lower lethal temperature of non-native blue tilapia (*Oreochromis aureus*) in the Susquehanna River of Pennsylvania (USA) was about 5°C. The local tilapia population overwintered in the thermal effluent of an electric power plant, so the plant temporarily lowered the water temperature during winter. This apparently eliminated the local population, but the tilapia persisted because of other thermal discharges along the river.

Complete dewatering to eradicate non-native fish populations has been proposed for some large reservoirs (CDFG 2007), but has largely been limited to small water bodies, usually aquaculture ponds (Alvarez *et al*. 2003; Mueller 2005). The water level of lakes or reservoirs is sometimes lowered in conjunction with the use of fish toxicants (CDFG 2007), thereby reducing the amount of toxicant needed and containing targeted fish within smaller and more exposed areas.

Increased harvest pressure as a method of controlling invasive or unwanted fishes can involve modification of regulations to promote angling, commercial harvesting or incorporating derbies and offering bounties (Lee 2001). However, because fishes vary in their susceptibility to capture, the methods used by anglers and commercial fishers are typically size and species selective. Consequently, the likelihood of removing an entire population through increased harvest is generally low (Thresher 1996, Yonekura *et al*. 2007).

Biological methods

The release of predators to prey on undesirable or invasive species as a form of biological control has a long history although it is not commonly used against invasive fishes. As in terrestrial environments, this approach to non-native fishes could have unintended consequences (Fuller *et al*. 1999). Contagious diseases such as koi herpes virus or KHV has potential use against non-native species, but is controversial because of potential harm to related desirable species (Gilligan and Rayner 2007) and likely difficulties with correcting unintended consequences. Moreover, surviving fish might have immunity to the disease, rendering the method useless after one application. Still, introduction of a highly-specific contagious disease could be helpful if combined with other methods.

Genetic manipulations which have been proposed include: 1) chromosome set manipulations involving production and release of triploid sterile non-native fish with the intent of reducing the population size of targeted naturalised individuals; and 2) recombinant DNA methods involving transgenic techniques designed to produce sterile fish or spread deleterious transgenes (i.e., "Trojan horse" genes) to a target non-native species (Gilligan and Rayner 2007; Thresher 2008). In Australia, there have been investigations into the use of "daughterless genetic technology" to combat introduced fish, especially common carp. This involves creating a heritable gene that suppresses the production of female offspring, causing a reduction in the nuisance population over successive generations (Gilligan and Rayner 2007). Few genetic manipulations have been tested in the field. One exception is release of sterile males to help control sea lamprey (*Petromyzon marinus*) in the Laurentian Great Lakes (Bergstedt and Twohey 2007).

A promising and potentially benign biological control method under development is to use pheromones, which are natural chemicals secreted by many fish and important in influencing their behaviour. To date, development of this method has been directed at the control of sea lamprey in North America (Sorensen and Hoye 2007). Field tests demonstrated that pheromone signals attract sea lampreys into traps. However, the campaign to control sea lamprey in the Laurentian Great Lakes—although providing ground-breaking methods of potential benefit for eradication of other species—has been intensive, long (over five decades), and expensive (approximately US$20 million annually) (Kolar *et al*. 2010).

CONCLUSIONS

Because invasive species cause ecological and economic harm, eradication remains an important management option. However, like other invasive animals and plants, invasive fishes can be difficult and expensive to eradicate. On islands, eradications of invasive fish may be simpler than in mainland areas, partly because an invading population is more spatially restricted. To date, the methods used against invasive fishes on Pacific islands are similar to those used elsewhere in the world. On the other hand, the state of knowledge on fish eradication is dynamic and, because each eradication project has its own unique set of problems, solutions may be site or species specific.

Eradication projects targeting invasive fishes are often controversial, partly because of the likelihood of collateral damage to native species (Britton *et al*. 2008) and especially when non-specific fish toxicants, such as rotenone, remain one of the few effective tools. The risk that an eradication attempt will harm native species is of particular concern on Pacific islands where native faunas include many endemic species. Consequently, early planning requires risk assessments to determine the relative benefits of eradicating non-native species against the potential harm native organisms. Such decisions need to be judged on a case-by-case basis, requiring awareness of the different eradication methods and strategies, and associated positive and negative consequences, as well as substantial knowledge of the targeted species, the invaded habitat, and substantial information on the native fauna present.

The time and effort expended on basic information about invasive fish depends on characteristics of the species, size and complexity of the invaded environment, risks that the population will rapidly or easily spread, and its potential undesirable effects. The possibility of eradication decreases and the potential costs increase as the invading populations disperse. Consequently, eradication is best attempted almost immediately upon discovery of new invasive populations (Simberloff 2009). Unfortunately, since monitoring is often inadequate, non-native populations are often large and widely distributed when biologists become aware of their existence.

Recognising the risks of delay, McDowall (2004a) concluded "…where potentially invasive species are known to be present, the first action must be to attempt control or eradication, and once that has been done, to then take the time to carefully evaluate the risk posed by a species." Similarly, Simberloff (2009) argued that successful eradication calls for quick action—in some situations a "scorched-earth" approach—with minimal time spent conducting research, although he recognised that some cases require sophisticated scientific research prior to action. For non-native fishes, a basic understanding of their biology is necessary to ensure that eradication methods chosen are appropriate and offer the greatest chances of success.

Successful eradications have key elements in common (Simberloff 2009): 1) detecting an invasion early and acting quickly to eradicate it; 2) sufficient resources allocated to the project from start to finish including post-eradication surveys and follow-up, if necessary; 3) a person or agency with the authority to enforce cooperation; 4) the targeted species studied well enough to suggest vulnerabilities (often basic natural history suffices); and 5) optimistic, persistent, and resilient project leaders.

Globally, improved methods and strategies are needed to eradicate invasive fishes, especially where these species are causing the decline of endemic or imperilled native fauna. Future research will likely focus on the control or eradication of a few of the more notorious invaders, although methods developed against one species may be applied to other taxa. Future needs include: 1) re-examination and adjustment of methodologies; 2) development and testing of additional ichthyocides especially those that are more selective and less harmful to non-target species; 3) newer biological techniques, including "Trojan genes" and pheromones, which should enable selective targeting of fish for removal. Unfortunately, it is likely that many of these advances will be costly to develop and field applications possibly decades away.

Because budgets are usually limited, setting priorities is essential. Focus is often directed at species perceived to be especially harmful. In considering Pacific islands, a complementary approach to species targeting is ecosystem prioritisation (Jenkins *et al*. 2009). This strategy recognises that a common goal of non-native eradication is protection of native biodiversity. Most native freshwater fishes inhabiting Pacific islands have complex life cycles and their survival is dependent on high connectivity between terrestrial, freshwater, and marine systems. To maintain biodiversity and reduce the impact of invasives, ecosystem prioritisation demands conservation and management of entire catchments, particularly those that are intact and unique (Jenkins *et al*. 2009).

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Argentine ants on Santa Cruz Island, California: conservation issues and management options

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Abstract Non-native Argentine ants (*Linepithema humile*) have established on California's Santa Cruz Island but are apparently not yet widespread. Santa Cruz Island is a highly valued conservation area, harbouring a large number of endemic plants and animals, and communities now rare in mainland California. Following eradication of non-native vertebrates and honeybees, Argentine ants were regarded as one of the major remaining threats to native biodiversity on the island. They were first detected at a single site in 1996 and were found at two more by 2004. Surveys in 2009 and 2010 showed that they had spread to three small sites along the lower reaches of the Island's largest waterway. They were not detected at 15 other sites on the island with heavy human use, or at 12 beaches used by recreational boaters. In 2009, we convened an expert working group which confirmed that Argentine ants are a serious threat to the island's biodiversity and recommended a management framework to: 1) detect and delimit *L. humile*; 2) implement biosecurity measures and engage island residents and visitors to prevent further spread and re-invasion; and, 3) contain and reduce existing infestations using toxic baits. In response, we launched a management framework with the goal of containing and perhaps ultimately eradicating this notorious invader from the island.

Keywords: Biodiversity conservation, eradication, feasibility assessment, island restoration, *Linepithema humile*

INTRODUCTION

There are approximately 20,000 species of ants on Earth (Ward 2006) but only about 150 species, fewer than 1% of the total, are known to have established outside their native ranges after dispersal by humans (McGlynn 1999). Most of the introduced species are restricted to human-modified habitats, but a small subset has spread into natural environments where they have significant and sometimes disproportionate negative effects on native biological diversity. Six species of ants are particularly widespread, abundant, and damaging worldwide (Holway *et al*. 2002), including the Argentine ant (*Linepithema humile*). Many studies have documented harmful effects of Argentine ants on native ants, other invertebrates, some vertebrates, plants and plant communities, and in countries that include New Zealand (Ward and Harris 2005; Ward 2009), Australia (Walters and Mackay 2003; Rowles and O'Dowd 2007, 2009), South Africa (Buys 1987; Christian 2001), the Mediterranean region (Way *et al*. 1997; Quilichini and Debussche 2000), and in California (Ward 1987; Gambino 1990; Human and Gordon 1997; Sockman 1997; Holway 1998, 1999; Bolger *et al*. 2000; Fisher *et al*. 2002; Suarez and Case 2002; Suarez *et al*. 2000) where they are invasive in coastal areas and the Central Valley. As with other invaders, islands seem to have been particularly vulnerable to the harmful effects of invasive ants (O'Dowd *et al*. 2003; Lach and Hooper-Bui 2010). For example, Argentine ants have been particularly damaging on the island of Maui (Cole *et al*. 1992).

Santa Cruz Island (250 km^2) is approximately 40 km south of Santa Barbara, California and is the largest of the eight California Channel Islands. A highly valued conservation area, it falls entirely within Channel Islands National Park although just 24% of it is owned and managed by the U.S. National Park Service (NPS). The remaining 76% is owned and managed cooperatively by The Nature Conservancy (TNC), an international conservation organisation. The island harbours at least 21 endemic taxa of animals (insects, birds, mammals, and herptiles) and 8 endemic taxa of plants (Junak *et al*. 1995; Schoenherr *et al*. 1999). It also supports many other species found on some of the other Channel Islands but not on the mainland, as well as several plant communities now rare on the California mainland.

Argentine ants were first detected on Santa Cruz Island in 1996 (Calderwood *et al*. 1999) and were quickly recognised as a cause for concern. Today the ants are likely the most damaging invasive species remaining on the island. To assess the extent of the threat it poses, presence/absence surveys of Argentine ants were conducted around the three previously recorded infestations and other high human use sites on the island. We also convened an on-site meeting of 18 experts on Argentine ant ecology and control and conservation land management from the U.S. and New Zealand to advise us on a course of action. Here we present findings from those efforts.

HISTORY OF ISLAND MANGEMENT AND ARGENTINE ANT INVASION

The arthropod fauna of Santa Cruz Island is not fully known but includes at least 8 endemic insect taxa (Miller 1985; Schoenherr *et al*. 1999). More systematic collections of ants have revealed 32 native species. None of these are endemic although *Messor chamberlini* is relatively common on the island but quite restricted and rare on the California mainland (D. Holway, UC San Diego pers. comm.). Besides the Argentine ant, the only other non-native ant species on the island is *Cardiocondyla ectopia* (Wetterer *et al*. 2000); however, its known extent is small and it is currently not a species of concern.

Santa Cruz Island has a long record of human occupation and use, dating back at least 7000 years (Glasow 1980). By the time the effort to set the island aside as a conservation area began in the late 1970s, Santa Cruz harboured large populations of several non-native animals and many species of invasive plants. Particularly damaging were thousands of feral sheep (*Ovis aries*) and pigs (*Sus scrofa*) as well as smaller numbers of cattle (*Bos taurus*), all of which had been intentionally introduced in the mid-1800s (Junak *et al*. 1995). These ungulates severely damaged vegetation on the island, stripping much of the island almost completely of plant cover, leading to severe erosion (Brumbaugh *et al*. 1982).

By 2006, NPS and TNC successfully eliminated cattle, feral sheep and feral pigs, which allowed a spectacular recovery of native vegetation in many areas (Junak *et al*. 1995; Morrison 2007). Feral honeybees (*Apis mellifera*) were also eliminated from the island by the early 2000s, reducing threats to native bees as well as pollination services provided to invasive plants favoured by honeybees (Wenner and Thorp 1994; Barthell *et al*. 2001).

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Argentine ants were first detected on Santa Cruz Island in 1996 (Calderwood *et al*. 1999) and were quickly recognised as a cause for concern. In 1997 and 1998, delimitation surveys were conducted around areas where Argentine ants had been detected and in other locations with a recent history of human use and movement of goods (Calderwood *et al*. 1999). Infestations were found at two sites, one covering approximately 1.5 km^2 (hereafter referred to as the Valley Anchorage site) and the other covering 0.04 km^2 (hereafter, the Blue site). The larger infestation was of a size consistent with arrival of the ants 5 to 10 years before the surveys were conducted (Calderwood *et al*. 1999). Both infestations appeared to radiate from sites of U.S. Navy installations dismantled in 1995, which led Calderwood *et al*. (1999) to speculate that heavy equipment transported to the island for use in these installations may have carried Argentine ants. The surveys also found that native ants were largely absent wherever Argentine ants were established (Wetterer *et al*. 2000, 2001). Unfortunately, no control action was taken at that time, in part because problems caused by invasive feral pigs (*Sus scrofa*) were severe and given higher priority. Both sites thus remained infested (Fig. 1).

There are two building complexes in the central valley of the island: a historic ranch compound that serves as the island headquarters for The Nature Conservancy, and a nearby University of California (UC) Field Station. Searches around both complexes failed to detect Argentine ants in 1997 and 1998, but a third infestation was found around the Field Station in 2004 (L. Laughrin, UC Natural Reserve System pers. comm.; Fig. 1). At this time, however, a feral pig eradication programme and efforts to recover the endangered island fox (*Urocyon littoralis santacruzae*) were in full swing, and took priority over addressing the Argentine ant invasion. With the recent completion of the feral pig eradication programme (Morrison 2007) and the island fox showing strong signs of recovery (Coonan and Schwemm 2008), managers have the capacity to focus on other management priorities.

METHODS

Argentine ant surveys

Surveys of the three known Argentine ant infestations, plus 15 other sites with high human use on the island, were conducted between 20 May and 16 June 2009. Non-toxic baits were placed at a total of 468 bait stations at the 18 sites. Bait stations were placed around and up to 600 m beyond the boundaries identified in previous surveys for the three known infestations (1997 and 1998 for the Valley Anchorage and Blue sites; 2004 for the UC Field Station site). These distances were based on Argentine ant invasion expansion rates of 10 to 100 m/yr recorded elsewhere in California (Holway 1998). Each bait station consisted of a 3 x 5 inch (7.6 x 12.7 cm) paper card with a few drops of organic maple syrup and several small pecan (*Carya illinoinensis*) nut pieces. Bait stations were revisited 30-120 minutes after they were set out, and the presence and quantity (1, 2-10, >10, >100) of Argentine ants on or immediately adjacent to the stations was recorded (Coastal Restoration Consultants 2009). Other ants on or immediately adjacent to the stations were identified to genus where possible and recorded (data not reported here).

On 13-15 November 2009 and 21-24 June 2010, more detailed surveys were conducted in the Cañada del Medio and Cañada del Puerto drainage and riparian areas, i.e. from approximately 100 m upstream of the Field Station, to the drainage's mouth at Prisoner's Harbor, approximately 5.5 km downstream of the Field Station. This catchment was judged most likely to harbour additional infestations since it is downstream of the Field Station infestation, and is used as the primary transportation corridor from the island's north coast to its interior. The surveys used a sucrose-water mix or a sweet gel (the non-toxic attractant in Xstinguish Ant Bait produced by Bait Technology of New Zealand; www.flybusters.co.nz/Bait+Technology.html) placed in small plastic vials as baits. Bait vials were placed at intervals of 15 m or less, and retrieved after 5 and 24 hours. Ants in the vials were identified and recorded. During the November 2009 and June 2010 surveys, entomologists familiar with Argentine ants and native ants of southern California also searched on foot through the vegetation and cobbles in the watercourse and associated floodplain and recorded the locations of any Argentine ants found.

A separate survey of twelve beaches commonly used as landing areas by recreational boaters was conducted in June 2010. This survey also used the non-toxic attractant in Xstinguish Ant Bait placed in small plastic vials. The baited vials were placed >15 m inland of the high water mark, and approximately 15 m from any other monitoring tube. Baits were left in place for 1 -3 nights then collected and the vials examined for the presence of Argentine ants.

Argentine ant management recommendations

We convened a three day meeting of an Expert Working Group in October 2009 on Santa Cruz Island, which included 18 experts in the ecology and control of Argentine

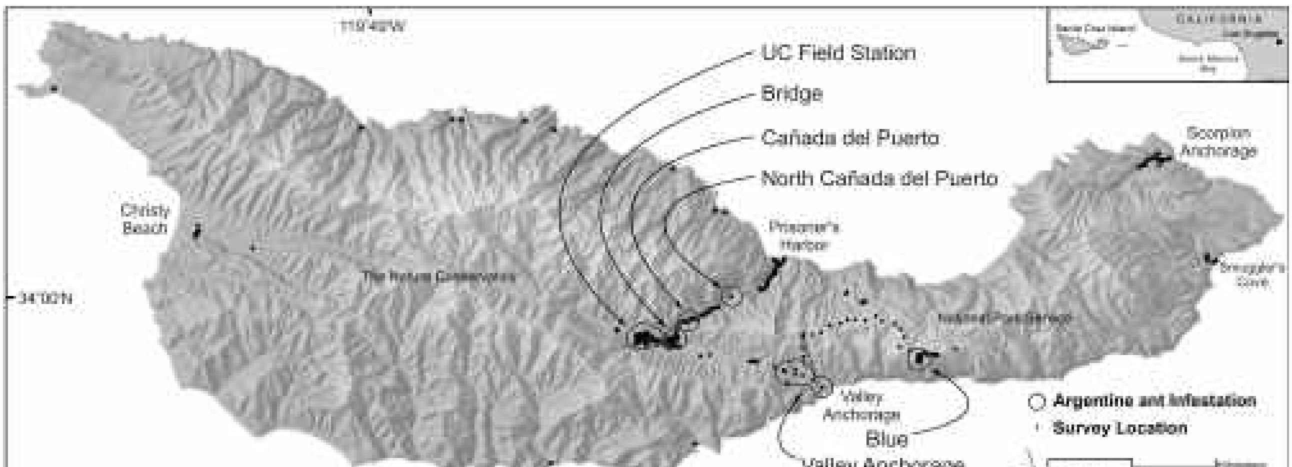

Fig. 1 Argentine ant infestations detected on Santa Cruz Island, California as of June, 2010 and sites where no Argentine ants were detected during 2009 and 2010 surveys. Inset shows the location of the Santa Cruz Island in California.

ants and conservation land management from the United States and New Zealand. The goals of the meeting were to: 1) describe known and potential impacts of Argentine ants on the island; 2) characterise management options; and, 3) make recommendations for management and monitoring.

RESULTS AND DISCUSSION

Surveys in 2009 and 2010

The survey conducted in May and June 2009 found continuing infestations at the three known sites. At Valley Anchorage and Field Station, Argentine ants had expanded in and near riparian corridors by rates of 30-60 meters per year since 1997. There was little or no evidence of expansion at the Blue site nor of expansion into dry uplands around any of the sites. No Argentine ants were found at any of the other 15 sites with heavy human use (Coastal Restoration Consultants 2009).

The November 2009 and June 2010 surveys detected three small infestations in the ephemeral stream drainage of Cañada del Puerto between the main ranch compound and Prisoners' Harbor (Fig. 1). One covered perhaps 75 m^2 in a wetland area north of the TNC ranch (Bridge site). Another measuring approximately 600 m^2 was found along the banks of an ephemeral waterway about a third of the distance between the ranch and Prisoners' Harbor (Cañada del Puerto site); and the third measuring approximately 300 m^2 was found further north along the ephemeral waterway close to Prisoners' Harbor (north Cañada del Puerto site). A total of six infestations are now known on the island.

No Argentine ants were detected in the June 2010 survey of the 12 beaches used as landing areas used by recreational boaters.

Recommendations from the Expert Working Group

The Expert Working Group reached consensus on three major points (numbered below). Each of the 18 group members was also asked to provide their personal conclusions and recommendations, which gave additional guidance on the three consensus points. We then summarised this input in order to outline the effort necessary to delimit, detect, prevent spread and re-invasion of existing infestations of Argentine ants, and to contain and ideally eradicate these populations. Details of the Expert Working Group's findings and recommendations, and our plans and actions to date to implement them are as follows.

1: Argentine ants are likely to spread and cause significant damage to the island's biological diversity, particularly to native ants and other arthropods.

Argentine ants are known to have harmful impacts on populations of native ants and other insects in coastal southern California and in other parts of the world. Typically, invasions by Argentine ants are followed by losses of medium- and large- bodied ants, plus reduced variation in dietary specialisation, behavioural repertoires and nest architecture, all of which are important to other plant and animal species (Ward 1987; Holway 1998; Holway *et al.* 2002). Several of Santa Cruz Island's 32 native ant species would be threatened if Argentine ants are not managed, including the relatively rare *Messor chamberlini* and *Pogonomyrmex californicus*, a harvester ant that collects and stores seeds. The ability of Argentine ants to displace and dominate other species is apparently greatly enhanced in California and some other regions they have invaded, in part because they show little or no aggression between nests and effectively form huge supercolonies with multiple queens. This apparently allows Argentine ants to maintain extremely high population densities (Suarez *et al.* 1999). The presence of Argentine ants has been associated with reduced abundances of flies, springtails, beetles, cynipid wasps, ticks, mites and spiders in northern and southern California (Human and Gordon 1997; Bolger *et al.* 2000) and of arthropods from eight orders on the Hawaiian island of Maui, including endemic spiders, moths, beetles, bees and flies (Cole *et al.* 1992). Argentine ants have been observed attacking yellow jacket (*Dolichovespula arenaria; Vespula germanica; V. pensylvanica*; and *V. vulgaris*) colonies in northern California (Gambino 1990) and were shown to compete successfully for nectar sought by honeybees (*Apis mellifera*) in South Africa (Buys 1987). Similarly, on Maui they entered the nests of a native solitary bee, *Hylaeus volcanica*, flushed out the adults and apparently preyed on larvae, none of which were found in areas infested by the ants (Cole *et al.* 1992). On the other hand, Holway (1998) found that while Argentine ants displaced all native ant species that feed above-ground except for the cold-tolerant, winter-active species *Prenolepis imparis*, they did not appear to affect the diversity or abundance of non-ant arthropods at his riparian woodland study site in inland northern California. Some studies (Cole *et al.* 1992; Human and Gordon 1997) also found that a variety of other non-native isopods and insects were actually more abundant in sites with established Argentine ant colonies, possibly examples of what has been termed "invasional meltdown" by Simberloff and Von Holle (1999).

Argentine ants also have direct and indirect negative effects on native vertebrates, including birds, mammals and reptiles as well as on plant-animal interactions such as pollination and seed dispersal that ultimately affect plant regeneration and community composition. Sockman (1997) found that Argentine ants were responsible for California gnatcatcher (*Polioptila californica*) nest failures. In coastal southern California, Argentine ants appear to negatively impact the coastal horned lizard (*Phrynosoma coronatum*) by displacing the native ant species the lizard prefers to eat and which support higher lizard growth rates (Suarez *et al.* 2000; Fisher *et al.* 2002; Suarez and Case 2002). Similarly, surveys at the Point Loma Ecological Reserve in San Diego found that Argentine ant density was negatively correlated with capture rates of lizards and salmanders (Atkinson *et al.* 2003).

In South Africa and southeastern Australia, invasive Argentine ants displaced native harvester ants and altered seed dispersal patterns in ways that may influence the species composition of native vegetation. For example, native large-bodied harvester ants, which preferentially gather seeds of large-seeded shrubs in the family Proteaceae, were absent from areas invaded by Argentine ants in the Cape Province of South Africa (Christian 2001). Other species of native ants that preferred seeds of smaller seeded shrub species in the same family persisted in invaded areas. As a result, the larger seeds were not dispersed in invaded areas and suffered very high rates of predation by native rodents, while the smaller seeds were dispersed and escaped predation. In turn, regeneration rates of the large seeded species were an order of magnitude lower in invaded areas than in uninvaded areas, while regeneration rates for the smaller-seeded species were not significantly different (Christian 2001). Near the coast southeast of Melbourne, Australia, Argentine ants displaced the native keystone disperser *Rhytidoponera victoriae*. They also dispersed significantly fewer seeds of a native *Acacia* but significantly more seeds of the non-native invasive shrub *Polygala myrtifolia*, another possible example of "invasional meltdown" (Rowles and O'Dowd 2009). Argentine ants may likewise be capable of displacing at least some of the Santa Cruz Island's native seed harvesting ants in the genera *Messor*, *Pheidole,* and *Pogonomyrmex*, and driving significant changes in seed dispersal (D. Holway, UC San Diego pers. comm.).

Argentine ants have great flexibility and capacity to exploit a wide variety of honeydew-producing aphids and scale insects (Choe and Rust 2006). In return for honeydew, Argentine ants protect the honeydew insects from predators which allows their densities to increase (Barzman and Daane 2001; Grover *et al*. 2008). Their partnership with honeydew insects gives them access to carbohydrates which may facilitate their invasion of natural habitats (Rowles and Silverman 2009) and can help them to thrive when other foods are scarce making them more difficult to control, both in agricultural settings and in conservation areas.

The workshop concluded that Argentine ants on Santa Cruz Island will likely have similar impacts to those reported elsewhere if the species is not managed and allowed to spread.

2: Additional Argentine ant detection work should be carried out, particularly at sites with high levels of human use and along the island's major drainage.

The Expert Working Group agreed that the sampling methods used in the survey conducted in May and June 2009 may have failed to detect Argentine ants in some locations, and that the Canada del Puerto in particular should be re-surveyed. Most participants recommended that additional detection work be carried out with sweet liquid or gel attractant.

3: A management programme to prevent the spread and additional introductions of Argentine ants and to suppress or eradicate existing populations should be launched.

Because of the scattered nature of known infestations on Santa Cruz Island, the Expert Working Group concluded that Argentine ants could be eradicated, but only if targeted with a coordinated, multi-year control effort. Eradication is realistic because Argentine ant queens are flightless and so – unless transported mechanically, such as by a flood event or by humans – can only disperse and form new colonies by walking, which limits their rate of spread (Krushelnycky *et al*. 2004; Silverman and Brightwell 2008).

The largest Argentine ant eradication effort recorded to date was carried out in Western Australia from the mid-1950s to 1988 (Hoffmann *et al*. 2010). The project was halted when organochlorine pesticides were banned and no effective alternative was found. The programme did reduce the area infested from about 18,000 ha to 1458 ha but failed to eradicate Argentine ants from the state. One reason for this failure was that some infested areas could not be treated due to agricultural and environmental concerns. Another contributor to failure was reduced public support as the infested area declined and fewer people had direct experience with the ants (Hoffmann *et al*. 2010). Argentine ant control and eradication have also been attempted on islands, including Maui in the state of Hawaii, Norfolk Island in the southwestern Pacific, and Tiritiri Matangi Island in New Zealand. Efforts to eradicate ants from Haleakala National Park and adjacent areas on Maui have not been successful and the ants continue to spread although efforts to control them around high value areas within the Park continue (Krushelnycky *et al*. 2005). Two rounds of baiting have been carried out on Norfolk Island but the effects of the second round have not yet been assessed and the overall effects of the project are not yet known (V. Van Dyk, Flybusters Antiants / FBA Consulting pers. comm.). The effort on Tiritiri Matangi has been more promising, and conservation managers believe that they are close to success (Ward 2009; C. Green, NZ DOC pers. comm.).

Based on experiences in other conservation areas, the Expert Working Group recommended a four-pronged approach:

Detect – Survey to detect any other Argentine ant infestations

Delimit – Use bait stations with non-toxic attractants to delimit all known infestations immediately prior to launching control efforts.

Control – Use toxic baits to reduce Argentine ant numbers and the area they infest with the goal of containing their spread or eliminating them from the island entirely.

Biosecurity – Implement protocols to prevent the spread of Argentine ants on the island and new invasions to the island.

Many Expert Working Group members underscored the importance of strong institutional commitment to the success of this approach, because effort and funding will need to be sustained over the long-term if containment or eradication of Argentine ants is to be achieved.

Argentine ant Management Framework

By July 2010, the following actions had been undertaken.

1. Surveys were carried out in November 2009 and June 2010 to detect any additional Argentine ant infestations in the island's largest drainage, the Cañada del Medio/Cañada del Puerto.

2. Another survey of twelve beaches was carried out in June and July 2010. Recreational boaters are known to land at these beaches and sometimes bring picnic and camping gear or other equipment which may harbour Argentine ants.

3. A delimitation survey to determine the spatial extent of all six known infestations was scheduled for September 2010. The results will inform management efforts to be carried out in 2011 and beyond.

4. Biosecurity efforts to prevent the spread of Argentine ants from known infestation sites or new introductions from the mainland have been launched. For example, because Argentine ants are known to be present within roughly 100m of a nursery for native plants on the island, all plant pots must be submerged in water and determined to be free of Argentine ants before they are allowed to leave the nursery. If Argentine ants are found in the nursery its operations will cease and all plants in it will be destroyed. We are in the process of developing a full biosecurity plan for the island. We will consult with biosecurity plans developed by the New Zealand Department of Conservation for Tiritiri Matangi Island and other islands, as well as the rodent prevention plan developed for the Pribilof and Aleutian Islands by the Alaska Department of Fish and Game (Fritts 2007). We are collaborating with graduate students at the UC Santa Barbara to evaluate potential vectors of new invaders, assess potential education and outreach measures for different island visitor groups (e.g., campers, researchers, boaters, day-hikers) and produce a cost-benefit analysis of different prevention and monitoring measures.

5. Efforts are underway to identify bait formulations (attractant plus toxicant) and baiting regimes which will kill Argentine ant queens and eliminate colonies and whose use on TNC and NPS properties will be permitted by state and federal regulatory agencies. In order to eliminate colonies rather than simply reduce the numbers of foragers, the effects of any toxicant we use must be delayed long enough to allow foraging workers to share it with workers who can pass it on to the queens they are tending (Rust *et al*. 2004). University of California Riverside researchers M. Rust and L. Greenberg are conducting laboratory and field studies to determine which attractants are most preferred and which toxicants (and at what concentrations) are most effective against Argentine ant queens. The attractants they

are testing include sucrose and water mixes and several commercially available gel and liquid formulations (minus toxicants). Their toxicant tests build on previous work by Roa (1992), Silverman and Roulston (2001), Klotz *et al*. (2004a; b) and Rust *et al*. (2004), and include the toxicants dinotefuran, fipronil, imidacloprid, indoxacarb, and thiamethoxam.

The product identification and permit process will take us past the period of the year when the launch of a control effort would be most effective in 2011. In the meantime, boric acid-based baits and botanical oils that kill ants on contact (thyme oil) are available and do not require permits. In situations where our 2010 surveys reveal a need to contain the leading edge of a known infestation, or where it may be possible to eliminate a small, newly detected infestation, we may choose to use these compounds in 2011.

Assuming that research, permitting, and other due diligence remains on course, we anticipate contracting for baiting to control Argentine ants starting in the (northern hemisphere) summer of 2012 or 2013. This will be followed by delimitation and control work in sequence each year until the delimitation data reveal that Argentine ants have been eradicated from the island or we determine that elimination or containment of Argentine ants will not be possible.

The infestation of Argentine ants at the UC Field Station poses the greatest concern for the unintentional spread of the ants. The Field Station is a hub of research activities on the island and it is feared that Argentine ants could be moved on vehicles or equipment based at the Field Station. We have therefore created a "quarantine zone" around the infestation with controls on the types of materials that can be moved out of the zone. Informational flyers are posted at the Field Station, and signage is posted on the periphery of quarantine zones. Additionally, in November 2009 we began deploying KM Ant Pro bait dispensers armed with dilute boric acid in a liquid sugar solution around the Field Station in order to reduce the number of foraging ants in the area.

CONCLUSION

It is not yet clear whether Argentine ants (*Linepithema humile*) on Santa Cruz Island can be eradicated or contained for the long-term. An Expert Working Group of experienced ant biologists, ant control specialists and conservation land managers concluded in 2009 that the damage these ants could cause if allowed to spread warrants a full effort to dramatically reduce their abundance and extent to prevent their spread to new areas, and if possible to eliminate them from the island. In response the Conservancy and the National Park Service launched a management framework that includes collaborating with researchers from the University of California and other institutions to identify control methods that kill Argentine ant queens and eliminate colonies. Assuming this research, as well as permitting and other due diligence remains on course, we anticipate starting full scale baiting to control Argentine ants in the (northern hemisphere) summer of 2012 or 2013. The ultimate goal of all these efforts is to protect native ants, other arthropods, and other native species threatened by Argentine ants.

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Strategies to improve landscape scale management of mink populations in the west coast of Scotland: lessons learned from the Uists 2001-2006

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Abstract Phase One of the Hebridean Mink Project (HMP) ran from 2001-2006 at a cost of £1.6 million and successfully eradicated invasive mink (*Neovison vison*) from 1100 km^2 of the southern islands of the Hebridean Archipelago, North Uist, Benbecula and South Uist (The Uists). Mink were also heavily controlled in South Harris to the North of the Uists to prevent reinvasion. A total of 532 mink were removed, and no further animals were caught or recorded in the eradication area in the last six months of the project. The project is now in its second phase and is continuing to remove mink from the remainder of the Outer Hebrides using lessons learned from the original eradication. This paper outlines the strategies developed in the first phase of the HMP. The strategies involved were: logistical, such as trap design and staff training; and ecological, such as using information on the behaviour of the population in space and time to effectively allocate resources.

Keywords: *Neovison vison,* eradication, Hebrides, adaptive resource management

INTRODUCTION

Invasive alien species (IAS) are currently listed as one of the greatest threats to global biodiversity, along with hunting and habitat loss (Atkinson 1996; Diamond 1984; Vitousek *et al.* 1997). They often prey on, compete with, or spread diseases, to native species. This is particularly true on offshore islands, where ecosystems tend to be impoverished; populated with less stable and more vulnerable restricted range species (Cronk 1997; Simberloff 2000).

The American mink (*Neovison vison*) is listed as one of the world's worst 100 IAS by the IUCN's Invasive species Specialist Group (www.issg.org). Mink now have a wide invasive range established as a result of deliberate or accidental releases from fur farms (Fig. 1; Bonesi and Palazon 2007; Dunstone 1993). The species can achieve high population densities, and has major impacts on native fauna, such as ground nesting birds. In continental Europe, mink have negative effects on indigenous European mink (*Mustela lutreola*) through direct interspecific competition including direct aggression (Sidorovich *et al.* 1999), and they have been implicated in the local extinction of water voles (*Arvicola amphibius*) in Great Britain (Strachan and Jefferies 1993). All countries of the European Union have international obligations to protected birds and habitats in Special Protected Areas (SPA) and Special Areas of Conservation (SAC) designated under the EU Birds and EU Habitats Directives. The directives were developed in response to the Ramsar Convention (1994) and Berne Convention (1979) to protect wildlife and habitats, and the Bonn Convention to protect migratory species (1980). Because of the effects of mink, their control or eradication is required in areas where these directives apply.

Feral mink populations established on the Western Isles of Scotland (Hebrides) after escaping or being deliberately released from two fur farms at Carloway on the Isle of Lewis in the 1950s (Angus 1993; Cuthbert 1973). The mink have since spread southwards through Harris. Attempts were made to stop mink from colonising the Uists (North and South Uists and Benbecula) (Angus 1993), but they successfully established feral populations across the entire archipelago within 40 years, most recently on South Uist in 2002.

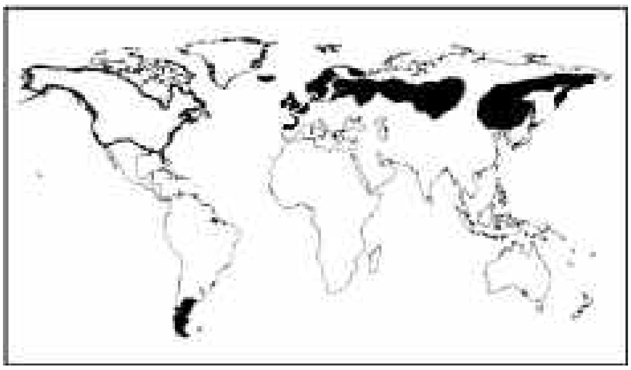

Fig. 1 The invasive distribution of mink (dark grey), from Dunstone (1993). Mink are native to Canada and North America.

On the Western Isles, mink have had severe effects on populations of fish (Bilsby 1999; 2001) and ground-nesting birds (Clode and MacDonald 2002). As up to £30 million of the Western Isles economy is based on tourism, with a large proportion of that based on wildlife tourism, hunting and fishing, mink potentially have an important economic as well as an ecological impact on the islands (Areal and Roy 2009; Moore *et al*. 2003; Roy 2006).

In this paper, I describe the history of an eradication programme against mink on the Outer Hebrides Islands, review the strategies applied and identify those that led to a successful eradication. Since the purpose of the paper is to demonstrate the lessons learnt, detailed analysis is only provided for those results that highlighted important strategic developments as the eradication progressed. These key developments enabled continual refinement of techniques that resulted in the elimination of populations of mink on large inhabited islands throughout the eradication are without detrimental effects on native populations of mammals.

THE HEBRIDEAN MINK PROJECT 2001-2006

The first phase of Hebridean Mink Project (HMP) ran from 2001-2006 (Roy 2006), and aimed to protect ground nesting bird colonies by: (i) eradicating mink (http://www.jncc.gov.uk/ProtectedSites/SACselection), in the Uists and (ii) reducing South Harris populations to prevent recolonisation over a total area of 1100km^2 (Fig. 2). The project was also acted as a pilot study for an island wide eradication campaign and was supported by a PhD research project (Helyar 2005).

The main method of removal was through live trapping and dispatch. Although the use of lethal traps is legal in the UK, these need to be checked daily. Furthermore, compared with live traps, lethal traps are more expensive, require more maintenance, more time and more skill to operate. We thus concluded that lethal trapping would not have saved time in this project.

The live trapping was supplemented with dogs, which searched for female mink in dens. Dogs were also used throughout the year as part of a mink monitoring campaign

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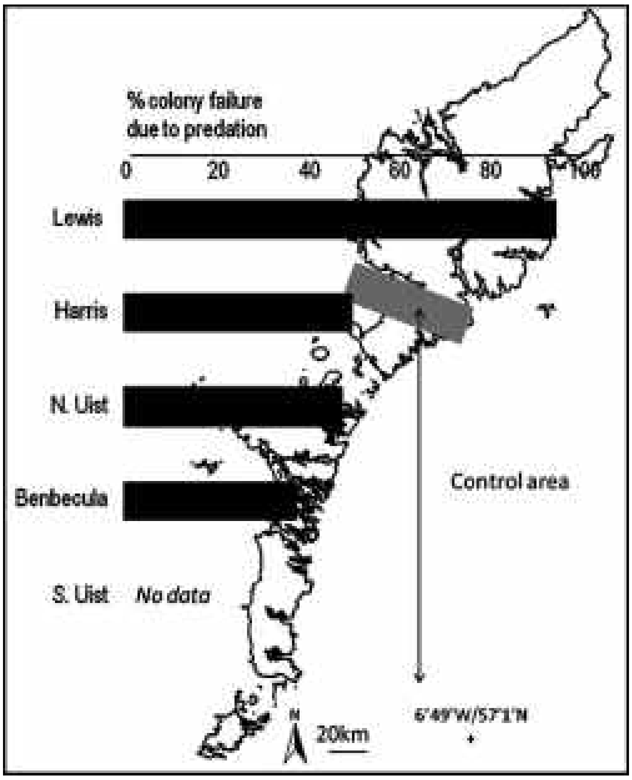

Fig. 2 The area over which mink were eradicated in the Uists, the area over which mink were controlled (Harris, North Uist, Benbecula and South Uist), and the area where mink ecology was studied as part of a PhD study (grey box)(Helyar 2005). Overlaid onto this is a comparison of the failure rates of tern colonies (*Sterna paradisaea, S. hirundo, S. albifrons)* within and outside of the control area 2004-2005.

within the control areas. The time devoted to searches with dogs was not recorded as a quantifiable measure of mink presence or absence (e.g., Theobald and Coad 2002), because dogs, dog handlers and search conditions varied throughout the project. However, sighting records were collated throughout the project and were weighted according to the member of the public making the report (Birks *et al*. 2004; Proulx *et al*. 1997).

Four and a half thousand traps approximately 400 m apart (actual distances ranged from 380-510 m) were entrenched into the ground along the coast and along the edge of inland waterways. The performance of each trap was monitored for the duration of the project. All trappers were involved in establishing trap lines in the first three months of the project throughout the control area on a zone-by-zone basis until traps covered the entire area. Thus trap lines and zones were not trapper specific. Once established, traps were only opened and set in coordination with the overall trapping programme, which usually lasted for a two weeks. Otherwise the traps were left unset to prevent accidental capture, until they were revisited later in the year. Most traps were revisited four to five times a year. When open, traps were checked daily; each trapper checked 30-50 traps a day. The project had a total of eight long-term trappers, with extra staff drafted in to assist during those seasons when mink are more mobile and easier to catch. In total, traps were opened for approximately 200,000 trap nights over the five year duration of the project. Traps were baited with fish in the first year of the project, but subsequent work showed that traps baited with commercially purchased mink scent gland (Kishel Scents and Lures, Saxonburg, USA) had significantly higher capture rates. As mink in traps rarely consume baits, all traps were baited with scent gland. Once caught, mink were humanely dispatched using hand held 0.22 calibre air pistols (J. Graham and Co. Inverness). The mink were aged as kits, juveniles or adults from tooth-wear, and sexed (Helyar 2005). Feral ferrets (*Mustela furo*) and rats (*Rattus* spp.) that were caught were also dispatched.

Sea bird colonies, in particular those of terns and gulls, were monitored annually during their breeding season, within and outside of the control area from 2002-2006. Data were gathered on productivity, hatching success, and nest failure (Ratcliffe *et al*. 2008; Roy *et al*. 2006).

Results 2001-2006

A total of 532 mink were removed from the control area (Table 1), with catch/ trapnight ranging from 0.015 to 0.0008 animals/trapnight/10km^2. The last mink was captured on the Uists in March 2005, with no further animals caught or detected for the remainder of the project, which ended in April 2006. The associated monitoring of tern colonies has also showed lower rates of predation-related failure (Fig. 2). Predation on tern colonies may also be by otters (*Lutra lutra*) and feral ferrets, which confound these data. These analyses assumed that the densities of the other predators, such as otters that prey on terns, remained constant throughout the project (Strachan 2006).

Table 1 Mink numbers caught on Harris and the Uists over the entire project lifespan.

	Harris	Uists	Total
Male	162	93	255
Female	131	117	248
Unknown	9	20	29
Total	302	230	532

Strategies developed and lessons learned

The eradication of mink was conducted within tight budgetary and time constraints, which required the development and implementation of logistical and ecological strategic guidelines. Here strategy has been broadly defined as the application of resources in space and time to maximise outcomes.

Logistical strategies

The greatest efficiencies were obtained from equipment and staff following an analysis of two areas: trap design and the skill of the trapper.

Trap design

The trap design selected had solid metal doors that were reliably visible with binoculars from 100m distance (Fig. 3). This meant that once set, traps could be checked without the need to approach the trap front. This minimised

Fig. 3 Buried traps in the Hebridean mink project (Photo S. Roy). This highlights the difficulty in seeing traps from a distance and the importance of the solid metal door.

trap disturbance. Also, being highly visible, a large number of supplementary traps could be set by the roadside and checked while trappers were driving to and from “walking traplines”.

It was estimated that when walking formal traplines the visible metal doors saved approximately 2-5 minutes in checking a trap (pers. obs.). Eight trappers were able to check 40 traps/day, only having to walk up to traps to set them on a Monday and close them on a Saturday. By checking the traps from a distance for the remaining four working days in a week (unless something was caught), there was the potential to save between 5.3 and 13.33 hours a week. Formal traplines were operated throughout the year with the exception of a 16-week period when animals were denning. If a trapper works for 49 weeks a year (excluding holidays approximately 1800 hours a year), the time saving over a year could potentially amount to 176-440 trapper-hours. This time could be redirected to check more traps, or carry out other tasks. In financial terms, if a trapper earns approximately £7.5/hour, this time amounts to £1300-£3300 a year, which could be used to purchase a further 120-300 traps.

Trapper skill

Though often widely spoken of, the skill of a trapper in catching animals is hard to quantify. In the HMP, success rates for each of the eight core trapping staff were assessed over the lifetime of the project, with dramatic results. It should be noted that all eight staff had equal access to trap lines and trap areas as they were established in the first three months of the project. Also all core trapping staff were able to tweak and modify traplines throughout the project.

Catch rates in traps set by different trappers showed great variation, with some trappers better at placing and setting mink traps than others (Fig. 4). When investigated further, the most successful traps were found to be operated by trappers three, four, and six. These were experienced gamekeepers and trapper four in particular had a long history of working on mink projects prior to this project. This information was later used to develop “quality assurance” roles for the most successful trappers, who regularly checked and tweaked trap lines and trained new trappers.

Ecological strategies

Ecological strategies were those developed to capitalise on mink behaviour, seasonal changes in population movements, and the way mink used space (different habitats) throughout the year. Trapping regimes were modified to maximise capture rates as a result.

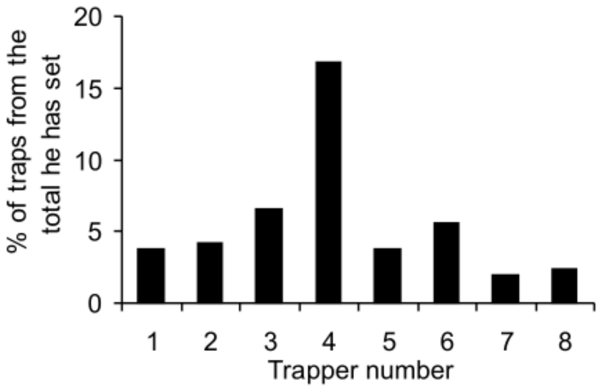

Fig. 4 The percentage of traps that have resulted in mink capture as set by trappers whose identities have been kept anonymous. The importance of experience is highlighted by game keeping experience (trapper 3, 4 and 6) and previous mink trapping experience (trapper 4) in trap performance.

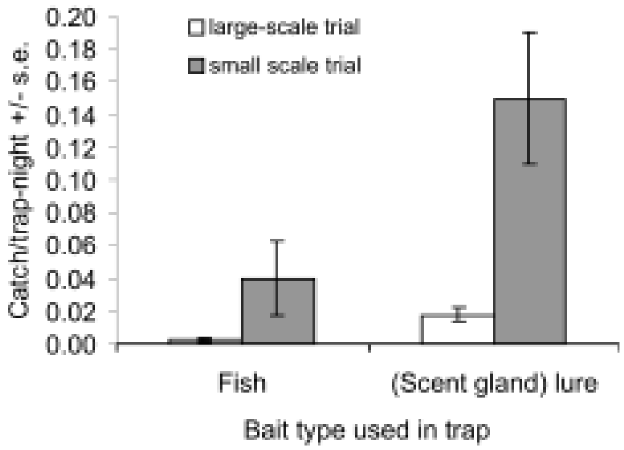

Fig. 5 A comparison of scent and fish baits in a small scale experiment over five small offshore islands, and a larger scale field trial in the Uists.

Mink behaviour

Like many small mustelids, mink use olfactory communication. For example, Roy *et al.* (2006) discuss in detail the effectiveness of mink scent glands to improve catch/unit effort. Traps baited with scent glands either extracted from culled animals or procured commercially (mink scent gland; Kishel Scents and Lures, Saxonburg, USA) provide a catch success an order of magnitude greater than traps using traditional fish baits (Fig. 5) There is also increasing anecdotal evidence that the use of predator scents may reduce the capture of non-target species (I. Macleod, Hebridean Mink Project Phase 2 pers. comm.). The use of scent-based lures thus had the advantage of leaving a greater proportion of traps available for mink capture. It also remained effective for several days after baiting, while food based baits often decomposed.

Seasonal changes in population movements

Mink have well defined seasonal patterns of behaviour (Dunstone 1993). In the northern hemisphere: 1) they establish and defend territories from November to January; 2) mate from January to April; 3) females set up breeding dens and rear young from the end of April to early July; and 4) disperse from late July to October. The mink are highly mobile and trappable during the dispersal and territorial periods, while during the denning period they are sedentary and difficult to catch (Fig. 6). In the HMP, this variability was exploited by drafting in extra staff and checking as many formal traplines as possible during the periods when the mink were mobile. During the denning period, nine trained dogs (spaniels) were used to locate den sites where females and young were subsequently trapped.

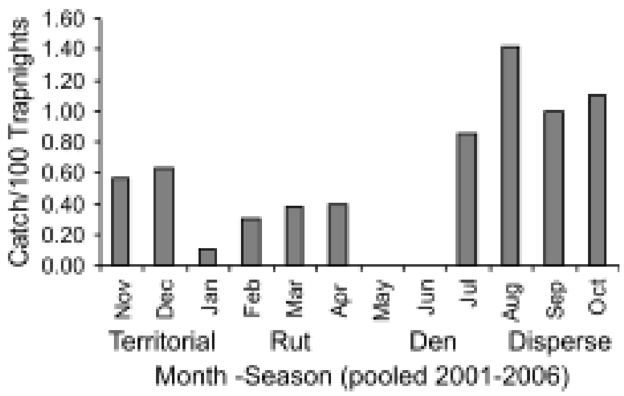

Fig. 6 The seasonal variation shown as catch/100 trapnights from November 2001 –July 2006 in Harris, and the Uists in the Hebridean Mink Project.

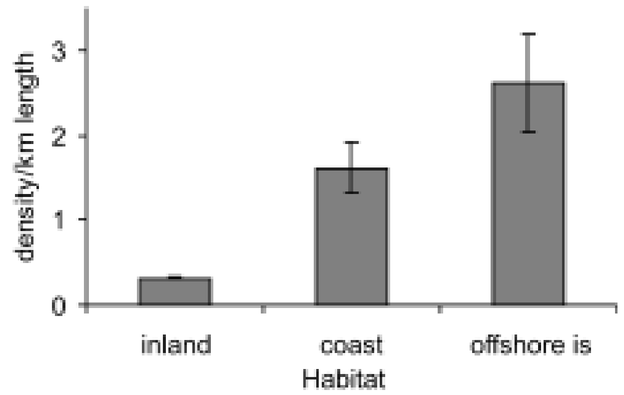

Fig. 7 Density estimates of mink in different habitat types in 2004 (Helyar 2005). Inland is defined by a habitat that is >200m from the sea. Density estimates for mink are often given in numbers/km length of riparian habitat due to their close association with water (Dunstone 1993).

A total of 11 active dens were found in 2004-2005, and these yielded 28 young and 10 adult females. Den sites were not excavated, because on the Hebrides mink re-use traditional den sites over several generations. Undamaged dens were used as a post eradication monitoring tool to ensure that no breeding mink were remaining on cleared areas (Helyar 2005).

The use of space by mink

Radio-tracking and capture-mark-recapture studies on a population of mink on Harris (Fig. 2;) showed that mink on the Hebrides are primarily coastal (Helyar 2005), with exceptionally high densities seen on offshore islands and the associated coastline (Fig. 7).

As a result of this information on spatial ecology of mink, a large number of previously untrapped offshore islands, including very small ones less than 1ha, were trapped and mink were successfully removed from many of them.

CONCLUSIONS

This project highlights the importance of applied research in developing project-specific strategies for large scale invasive species management programmes. Throughout its lifespan, regimes used in this project have evolved and been refined to great effect. Both the logistical and ecological data were collected, collated, combined and analysed to make informed decisions through a process of adaptive resource management. Such approaches become necessary when it is not always possible to undertake well-designed experiments due to time and financial constraints (Walters and Holling 1990). Applied information of the type needed by invasive species managers, information that combines ecological and logistical elements, and information on failures as well as successes, is not always readily available in the literature (Roy *et al.* 2009). The HMP succeeded because this information was recorded and used from the outset, having been collected from short, targeted research projects such as experimentation with scent glands, and from an applied PhD study associated with the project (Helyar 2005). Learning from the successes and failures from projects such as this one means operations can be scaled up more effectively to incorporate larger land areas and carry out eradications and control operations at ever increasing landscape scales.

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Rat invasion of Tetiaroa Atoll, French Polynesia

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Abstract All three species of invasive rats are found throughout the Pacific Ocean: *Rattus rattus, R. norvegicus* and *R. exulans*. Polynesians historically introduced *R. exulans*, after which competitively dominant *R. rattus* and *R. norvegicus* were introduced by Europeans. However, the competitive processes in island invasion among rats have never been well documented. Tetiaroa atoll, in the Society Islands, consists of 12 small coral islets ("motu") with remnant coconut plantations from the early 20th century. *Rattus exulans* was the only species present on the atoll until *R. rattus* was first documented in the 1970s. We review the history of Tetiaroa, and document the current extant distributions of *R. rattus*, *R. exulans* and the seabird community. Genetic studies confirm the species and locality of introduced rats with COI barcoding. Microsatellite analyses suggest recent isolation of the *R. exulans* populations on separate motu, whereas *R. rattus* on the north-west motu appear to be one meta-population. Colonies of small seabird species are generally associated with sandy areas on small motu with only *R. exulans* present. Only larger seabird species such as frigates and boobies successfully breed on motu with *R. rattus* present. With hotel development and pest control now under way, the challenge is to manage rat eradication and biosecurity measures both within and from outside of the atoll in coordination with preserving the seabird community. Studies such as this provide novel opportunities to understand competitive interactions between species.

Keywords: Biosecurity, competition, genetics, island, microsatellite, *Rattus*, seabirds

INTRODUCTION

The sequence of introductions (or assembly history) of introduced species can play an important role in their establishment and the final community composition where multiple invasive species interact (Drake 1990; Chase 2003; Courchamp *et al*. 2003). For example, where one invasive species is already established, the introduction of a second species can either exclude competitively inferior species or lead to changes in their abundance, behaviour, or trophic position (Grosholz 2005). Three species of the genus *Rattus* are widely distributed invasive pests (Amori and Clout 2003). Across 123 of the world's archipelagos, Pacific or Polynesian rats (*R. exulans*) are found on 24% (*n* = 30), brown or Norway rats (*R. norvegicus*) are found on 36% (*n* = 44) and black or ship rats (*R. rattus*) are found on 50% (*n* = 61) (data from Atkinson 1985). *Rattus exulans* is invasive throughout the Pacific Ocean, where it was introduced by Polynesian immigrants dispersing from south-east Asia over the last 3,500 years (Matisoo-Smith *et al*. 1998; Matisoo-Smith *et al*. 2009). The cosmopolitan invasive rats (*R. norvegicus* and *R. rattus*) did not reach islands in the Pacific until the arrival of European explorers 300 years ago, with a colonisation peak following World War II (Atkinson 1985). Upon arriving at islands already colonized by *R. exulans*, *R. norvegicus* and *R. rattus* competitively dominated (e.g., Baker 1946; Storer 1962; Williams 1972; Twibell 1973; Spennemann 1997; Russell and Clout 2004; Harper and Veitch 2006), although *R. exulans* may have resisted invasion on some islands due to an incumbent advantage (e.g., Roberts 1991; Russell and Clout 2004).

Identification of some species of *Rattus* can be difficult if based on morphological traits alone (Robins *et al*. 2007). *Rattus rattus* is a particularly problematic cryptic species 'complex', possibly comprising multiple species, sub-species and lineages (Aplin *et al*. 2003; Robins *et al*. 2007, 2008). Two different chromosomal forms are generally recognised, one Oceanian (2n = 38) and the other Asian (2n = 42) (Yosida *et al*. 1974; Baverstock *et al*. 1983). The Oceanian form (also known as European), named *R. rattus* by Musser and Carleton (1993), is generally the most invasive. However, the Asian form, named *R. tanezumi*

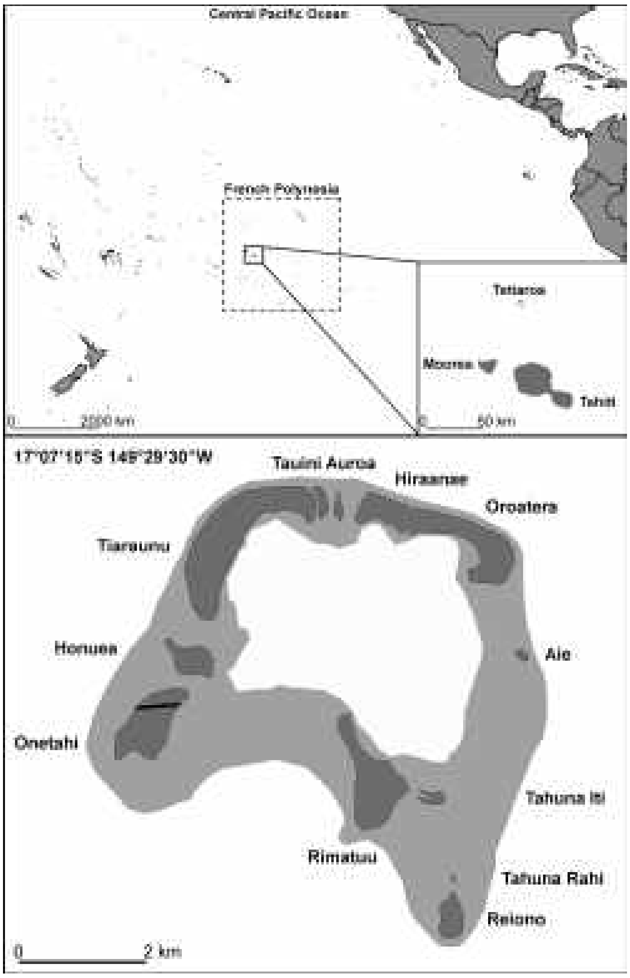

Fig. 1 Dark grey indicates land and light grey submerged coral reefs. Landing strip indicated on Onetahi. Tahuna Iti is colloquially named 'the bird island' («Ile aux Oiseaux»). Tahuna Iti and Tahuna Rahi have changed substantially over the past 50 years.

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by Musser and Carleton (1993), recently invaded McKean Island in the Phoenix Islands (Pierce *et al*. 2006). DNA barcoding based on mtDNA regions such as COI may provide more reliable species identification within the *R. rattus* complex (Robins *et al*. 2007).

Tetiaroa atoll (3366 ha; 17°07'15"S 149°29'30"W), lies 50 km north of Moorea and Tahiti in the Society Islands of French Polynesia (Fig. 1). Tetiaroa is one of 115 sites of important conservation value in French Polynesia (Meyer *et al*. 2005). The atoll comprises 12 low-lying vegetated coral islets, locally called "motu", and an emerging sandbank (Motu One) east of Tahuna Rahi, all roughly circling a large lagoon. Names of motu vary among reports due to typographic errors, and the changing geography of the landscape. Tahuna Rahi and Tahuna Iti have dramatically changed size and moved over the last 50 years (comparison to a 1955 aerial photo). Tahuna Rahi has decreased from 2.0 to 0.5 ha, while Tahuna Iti has increased from 5.2 to 5.9 ha, and more notably shifted from 350 m to less than 10 m offshore of Rimatuu. Tahuna Iti was presumably also the smaller island of the two historically, based on its name ('iti' translates as small). Most motu are dominated by abandoned coconut plantations formely exploited for copra (coconut oil), although Reiono retains substantial native vegetation dominated by a 20 m canopy of the tree *Pisonia grandis*. Archaeological sites from Polynesian settlement on Tetiaroa have been dated to 1500-1600 A.D., including pig remains (*Sus scrofa*) on Onetahi (Sinoto and McCoy 1974), although these early radiocarbon dates may be inaccurate (Spriggs and Anderson 1993). There has been short- and long-term human habitation since that time. Recent habitation, and hotel development on Onetahi, has led to further species introductions and vegetation alteration.

The archaeology was assessed in the early 1960s (exact date unrecorded) by Pierre Vérin, Raoul Teissier and Henri Picard (Teissier 1962, Vérin 1962) and in December 1972 by Yosihiko Sinoto and Patrick McCoy (Sinoto and McCoy 1974). The avifauna (predominantly seabirds) was assessed from 1972-1975 by Jean-Claude Thibault (Thibault 1976), and in 1992-1993 by Philippe Raust and Albert Varney (Raust and Varney 1992, Raust 1993). The ecology of the atoll, focusing on vegetation, was studied by Marie-Hélène Sachet and Francis Raymond Fosberg in 1973-1975 and 1982-1983, resulting in an exhaustive plant list for each motu (Sachet and Fosberg 1983). Further botanical visits and a revised plant list were made by Jean-François Butaud in 2003 and 2006 (Butaud 2006).

In the early 1960s, *R. exulans* were the only rats described on the atoll (Teissier 1962), but by the early 1970s there was a 'recent population explosion of a tree-dwelling rat' (Sachet and Fosberg 1983), and 'large sized rats' were seen under red-footed booby (*Sula sula*) colonies on the northern islands around 1972-1975 (Thibault 1976). Both observations are presumed to be of *R. rattus* and coincide with new ownership and development on Tetiaroa. Cats (*Felis catus*) were reportedly introduced to Tetiaroa to control abundant rats after 1904, but by the 1970s only remained on Onetahi (Thibault 1976). Fourteen semi-wild cats were removed from Onetahi in early 2009 by trapping (N. Leclerc pers. comm.). Rat eradication was also attempted commencing in June 2009 with a 50 m grid of bait stations and hand-spread Talon wax baits at a rate of approximately 10 kg/ha^{-1} over two sessions. After surviving rats were detected, a third follow-up application was made. Domestic pigs and dogs were probably also on Onetahi until recently (Sachet and Fosberg 1983), and a pair of dogs remain on Onetahi and regularly swim across to Honuea (pers. obs.). In this paper we record the distribution of *R. exulans* and *R. rattus* on Tetiaroa and describe how introduced rats interact with the extant seabird community. Genetic analyses are used to verify the species and population structure of invasive rats.

Table 1 Distribution of rat species on Tetiaroa. TN = trap nights.

Motu	Size	*R. rattus*	*R. exulans*	TN
Onetahi	73.8	✔	✔	30
Honuea	28.0	✔ *	✔	45
Tiaraunu	163.4	✔ *	✔ *	50
Tauini	6.7	✔ *	✔	10
Auroa	3.9	✔ *	✔	15
Hiraanae	34.0	✔ *	✔	25
Oroatera	81.4	✔	✔	10
Aie	2.4	–	✔	40
Reiono	21.4	–	✔ *	50
Tahuna Rahi	0.5	–	–	20
Tahuna Iti	5.9	–	✔	75
Rimatuu	88.3	–	✔ *	65
Total	509.7	7	11	435

* indicates samples from motu included in Structure genetic analysis

METHODS

In July 2009, we visited each motu (Table 1) and determined the species of rat present through a combination of observation and snap-trapping (Victor Professional) by the Société d'Ornithologie de Polynésie. Identification of rats in the field used morphological traits, particularly the dark stripe of fur on the outer hind feet, which is present on *R. exulans*, but absent from *R. rattus*. Sex, body-weight, head-body length, tail length and reproductive condition were all recorded, and a 5 mm tail or paw tissue sample stored in 70% ethanol for genetic analyses. In January 2010, motu where we had not previously trapped rats were revisited and ten waxtags (Pest Control Research) were placed overnight in order to verify previous negative trapping results.

Genomic DNA was extracted using a high salt extraction. Cytochrome C. oxidase subunit 1 (COI) was amplified using the following primers (Meyer 2004): dgLCO-1490 (5'-3') GGT CAA CAA ATC ATA AAG AYA TYG G, and dgHCO-2198 (5'-3') TAA ACT TCA GGG TGA CCA AAR AAY C. using standard PCR protocols. Six microsatellite markers characterised for *R. norvegicus* but suitable for other *Rattus* species were used: D19Mit2, D7Rat13, D15Rat77, D10Rat20, D20Rat46, D16Rat81 (Jacob *et al*. 1995). Each forward locus primer was tailed with M13 at the 5' end and a nested PCR was performed which included a fluorescent dye-labeled M13 primer (Schuelke 2000). PCR was performed in 10 µl volumes, containing 1 µg DNA, 0.1 µM of the M13-tagged primer, 0.1 µM of the other primer, and 0.1µM of the fluorescent dye-labeled M13 primer, and 0.2 µM of each dNTP, 1 unit *Taq* polymerase, and 1x reaction buffer with 1.5 mM $MgCl_2$. For each locus, annealing temperature was at 55°C for 30 cycles, followed by 10 cycles at 50°C to incorporate the fluorescent dye in the PCR product. PCR products were run on an ABI 3730 (Applied Biosystems). Amplification size was scored using Genemapper v.4.

We used Structure v.2.3.1 (Pritchard *et al*. 2000) to identify the number of clusters, *k*, for both species of *Rattus*. We only included motu where more than one rat was caught. We implemented the admixture model without priors of sampling location, with correlated allele frequencies and a burn-in of 50,000 and MCMC chain of 200,000, with five iterations for each of $k = 1, \ldots, m + 1$ (m = the total number of motu in the analysis). For each motu we used Arlequin v.3.1 (Excoffier *et al*. 2005) to estimate F_{ST} with 1000 permutations to estimate p-values. Microsatellite variation across the atoll was displayed visually with a principal components plot of the log posterior genotype probabilities (Russell et al. 2010). All motu with five or more captures were deemed reference populations, so each individual has a multi-dimensional coordinate consisting of its log posterior genotype probability for each reference population.

To investigate possible source populations for the recently arrived *R. rattus* we obtained tissue samples from four other major atolls in the Society Islands (atoll, sample size; Tahiti, 2; Moorea, 2; Huahine, 1; Raiatea 4) and compared these with four individuals from Tetiaroa. Complete cytochrome b (cyt b) was amplified using primers L14723 (5'-ACC AAT GAC ATG AAA AAT CAT CGT T-3') and H15915 (5'-TCT CCA TTT CTG GTT TAC AAG AC-3'), and in addition we amplified a further 758 bp at the 3' end of the cyt b gene comprising two tRNAs and a partial D-Loop region (Tollenaere *et al*. 2010). Polymerase chain reactions (PCR) were performed in a 25 µl total volume containing: 2 µl of extracted DNA, 0.5 µl of each primer (10 pm/ul), 200 µM of each dNTP, 1 µl BSA (10 mg/ml), and 1.25 U of FastStart Taq DNA Polymerase in the appropriate 1x Buffer with $MgCl_2$ (Roche Diagnostics). Samples were subjected to an initial denaturation at 95 ºC for 4 min, followed by 35 cycles of denaturation at 94 ºC for 45 s, annealing at 55 ºC for 45 s, and extension at 72 ºC for 1 min, with a final extension phase at 72 ºC for 10 min.

All Tetiaroa tissue specimens are lodged as part of the "Moorea Biocode project" (JR-2009-01 to JR-2009-78) (Check 2006).

Seabird distribution for each motu was determined from presence of the most abundant species, generally on the lagoon side, in July 2009.

RESULTS

Rats were detected on all motu except Tahuna Rahi (Table 1). On Aie and Tahuna Iti rats were not trapped but were subsequently verified at low density with waxtags and presumed to be *R. exulans* based on seabird presence and rats present on neighbouring motu. *Rattus exulans* inhabited all rat-invaded motu, whereas *R. rattus* were only found on the north-west chain. *Rattus exulans* were often observed throughout the day on motu with and without *R. rattus*, while *R. rattus* were never observed. Only one juvenile *R. exulans* was trapped on Onetahi as the concurrent rat eradication program during our trapping had substantially reduced rat numbers, while *R. rattus* were neither trapped nor observed on Onetahi but had been previously recorded. In January 2010, *R. rattus* and *R. exulans* were both widespread though not abundant on Onetahi. *R. exulans* were observed in abundance in the late afternoon on Hiraanae and Oroatera but were not trapped. Morphologically *R. exulans* were within the normal range but *R. rattus* were particularly large (Table 2). All three colour forms of *R. rattus* were found. Most rats caught were reproductively active adults, as indicated by enlarged testes in males and uterine scars and/or embryos in females. At least 18% of rats trapped were missing part of their tails, but with no clear pattern regarding sex or species.

Genetic samples were obtained from all rats trapped and were used to verify species and the extent of gene-flow across key water barriers (Fig. 2). COI barcoding results were compared to sequences of *Rattus* species available on Genebank. *Rattus exulans* on Tetiaroa aligned with those from the Pacific region (Robins *et al*. 2008), and *R. rattus* aligned with those from French Polynesia and the Pacific region (Robins *et al*. 2007). In our Structure analysis, we only included motu where more than one rat was caught (m = 3, n = 35 for *exulans*, m = 5, n = 36 for *rattus*). Our three sufficiently sampled *R. exulans* motu were isolated from one another (> 1.5 km), well outside the known swimming range of *R. exulans* (Russell *et al*. 2008). Structure found relatively equal support for k = 1 or 3. Support was marginally stronger for k = 3 but with much greater variances on estimated probabilities, which increasing simulation length did not alter. F_{st} values for *R. exulans* were significantly different among all three motu (Table 3). *Rattus rattus* were sufficiently sampled from five adjacent motu (< 500 m), predominantly around two major adjacent water-crossings, within the known swimming range of *R. rattus* (Russell *et al*. 2008). Structure found equal support for all of $k = 1,\ldots,6$. F_{st} values for *R. rattus* averaged less than 0.1 between motu, and were generally significantly different only between motu not adjacent to one another (results not shown). Allelic diversity was markedly different between the two species. Across the atoll, *R. exulans* loci

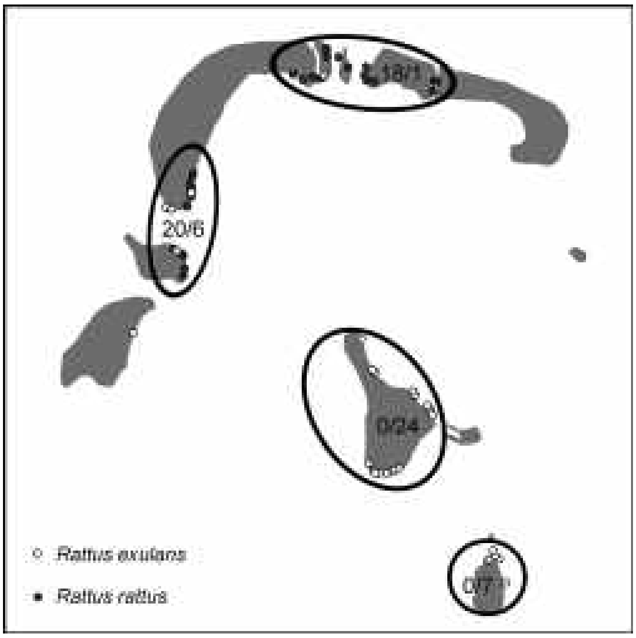

Fig. 2 Trapping locations and genetic sample sizes (excluding Onetahi) for rats on Tetiaroa (*R. rattus*/*R. exulans*).

Table 2 Average morphological measurements of adult *Rattus exulans* and *R. rattus* on Tetiaroa.

Species	Sex	n	Weight (g)	Head-body length (mm)	Tail length (mm)
R. exulans	M	10	77 (53-97)	150 (126-162)	163 (151-176)
	F	13	61 (48-70)	139 (114-149)	150 (132-163)
R. rattus	M	9	240 (200-308)	214 (202-228)	243 (215-268)
	F	14	192 (133-272)	206 (194-221)	233 (210-257)

Table 3 F_{st} values (3 d.p.) between Honuea, Reiono and Rimatuu for *Rattus exulans*.

F_{st}	Honuea	Reiono	Rimatuu
Honuea	0		
Reiono	0.120*	0	
Rimatuu	0.141*	0.167*	0

* significant at $p < 0.01$ (1000 permutations).

were characterised by long consecutive runs of two base pair microsatellite repeats, although any given motu would have a subset of these allele lengths. The mean number of alleles per locus globally was 9.2 (range 4 – 16). In contrast, across the atoll, *R rattus* loci were characterised by limited allelic diversity, and any given motu would include most of the globally available allelic diversity. The mean number of alleles per locus globally was 4.4 (range 3 – 6). Principal components analysis of microsatellite log posterior genotype frequencies among individuals supported our Structure results, with evidence of strong differentiation in *R. exulans* populations (Fig. 3a), but only weak differentiation in *R. rattus* populations (Fig. 3b). *Rattus rattus* did tend to align along the direction of their invasion front originating at Onetahi, with neighbouring motu at the lower right and more distant motu at the upper left (Fig. 3b), possibly coinciding with patterns in genotype frequency drift from serial founder events. Only one cyt b/D-loop haplotype was found among the 13 rats from five different atolls in the Society Islands (Genbank sequence HQ588111).

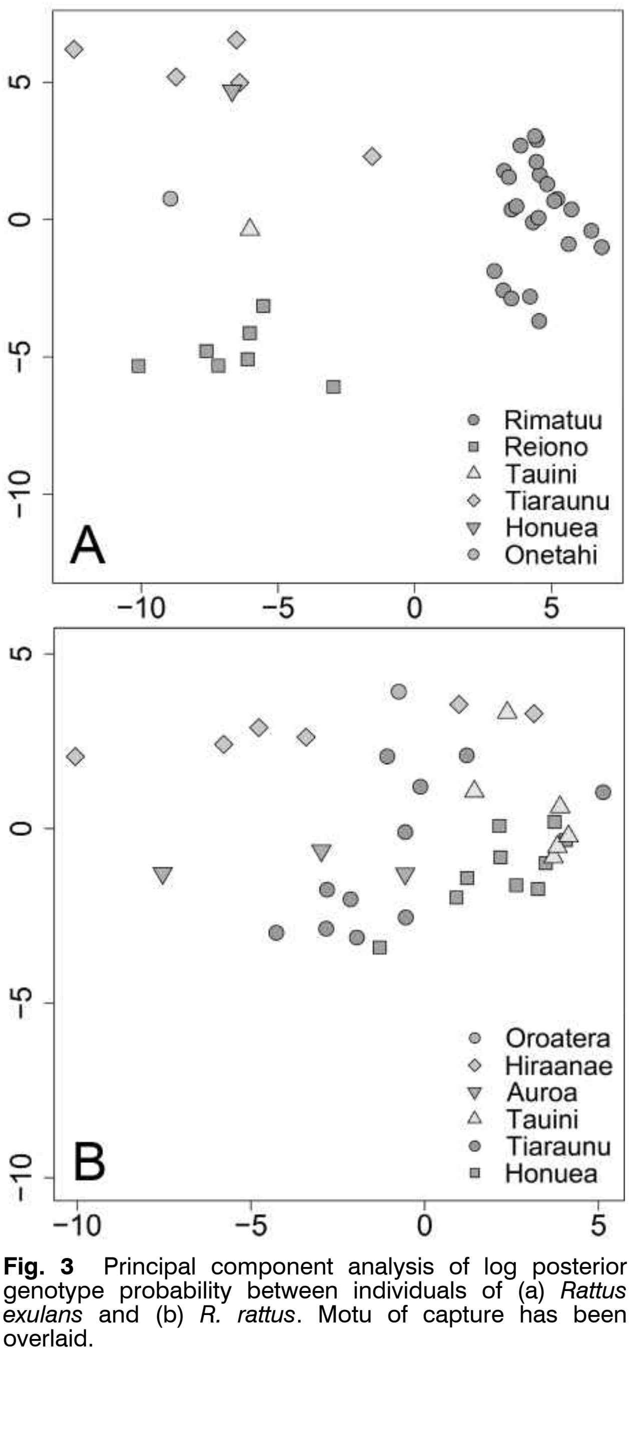

Fig. 3 Principal component analysis of log posterior genotype probability between individuals of (a) *Rattus exulans* and (b) *R. rattus*. Motu of capture has been overlaid.

Colonies of small seabirds such as noddies (*Anous stolidus*) and terns (*Onychoprion fuscatus* and *Thalasseus bergii*) were only found on small motu where *R. exulans* was the only species of rat present (Fig. 4). Larger seabirds such as frigatebirds (*Fregata minor* and *F. ariel*) and boobies (*Sula leucogaster* and *S. sula*) could breed in the presence of either species of rat (Fig. 4). For all seabirds, every reproductive stage (adults incubating eggs, juveniles and small chicks) was present, except for the small number of *Onychoprion lunatus* for which we only noted the presence of two juveniles. Since most of these species breed all-year round, numbers may differ at other times of the year.

DISCUSSION

Dominance of *R. exulans* by *R. rattus* has been widespread on islands of the Pacific (see Atkinson 1985). The relatively recent arrival of *R. rattus* on Tetiaroa provides an excellent opportunity to study how the process of domination proceeds. *Rattus rattus* successfully established in the presence of *R. exulans*, although how much of a detrimental effect this has had on incumbent *R. exulans* populations remains an open question. On Tetiaroa, *R. exulans* persist on even very small motu with *R. rattus*. In contrast, on McKean Island (49 ha) in the Phoenix Islands, a 2001 invasion of *R. tanezumi* appears to have completely replaced the incumbent population of *R. exulans* (Pierce *et al*. 2006). On Tetiaroa, the invasion by *R. rattus* over *R. exulans* has little positive benefits for the wider island community given that *R. rattus* is the more damaging invasive species (Jones *et al*. 2008). In New Zealand, *R.*

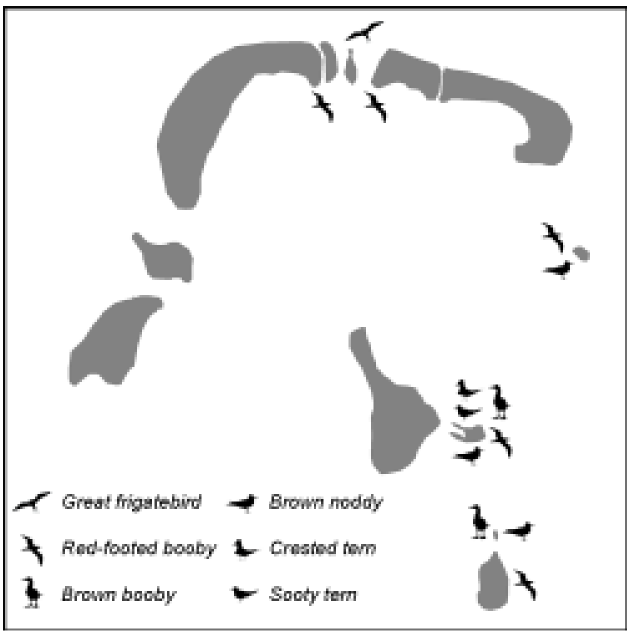

Fig. 4 Distribution of abundant seabirds on Tetiaroa (2009).

rattus dominates over *R. exulans* and populations only co-exist on islands larger than 100 ha (Russell and Clout 2004), although data are lacking for smaller islands, and mis-identification may be possible. In the tropics, however, introduced rats appear able to co-exist on smaller islands, and with less negative effect upon one another.

Our genetic results are constrained by small sample sizes, which limits our inferences. Nonetheless, the patterns of allelic diversity, sequential pair-wise mutations, and clustering in *R. exulans* are congruent with the hypothesis of a single historically large and diverse population (k = 1), either prior to introduction to Tetiaroa or on Tetiaroa but with regular gene-flow between motu. More recently, *R. exulans* on different motu have become isolated (k = 3), and the patterns of allelic diversity we observed are generated by a combination of genetic drift and our sample sizes, where in either case allelic diversity becomes a subset of the original global population. This change in dynamics is likely to have arisen when heavy use of the atoll by Polynesians ceased around the start of the 20th century. The limited allelic diversity in *R. rattus* suggests only a small number of founders, although probably more than one (the 'single pregnant female' hypothesis; Miller *et al*. 2010). Presuming *R. rattus* arrived in Onetahi and then subsequently invaded the north-west chain through isolated invasion events, we would expect to see a signature of sequential founder events (Clegg *et al*. 2002). However, given the rapid invasion of the entire north-west chain in the 1970s, only weak evidence for sequential founder events, and poor discrimination among the number of clusters, it is most likely that *R. rattus* form a single meta-population with regular gene-flow by swimming across the north-west chain. The entire north-west chain must be considered as a single eradication unit for *R. rattus* (Robertson and Gemmell 2004, Abdelkrim *et al*. 2005).

Despite a small channel crossing (tens of metres), *R. exulans* are apparently absent from Tahuna Rahi. This is likely a result of the complete inundation of the previously larger Tahuna Rahi prior to its reformation as the current nearby smaller motu (e.g., Sachet and Fosberg 1983). On Tahuna Rahi, the absence of rat gnaw on pandanus (*Pandanus tectorius*) and coconut (*Cocos nucifera*) nuts was a good indicator of rat absence, although it was not guaranteed when rats were also at low density such as on Aie or Tahuna Iti. *Rattus rattus* on Tetiaroa were particularly large, and with relatively short tails compared to body length. Both species of rat most likely benefit from the abundance of fallen coconuts that they open, and the presence of enhanced nutrient inputs under large seabird colonies.

Identifying a local source population for the recent *R. rattus* invasion of Tetiaroa was not possible due to a lack of haplotype variation among introduced *R. rattus* of the Society Islands. This lack of variation is most likely a consequence of the sequential invasion of *R. rattus* across the Pacific, meaning genetic diversity was already relatively homogeneous once *R. rattus* arrived in eastern-most French Polynesia. Populations of *R. rattus* in the Society Islands are likely to share a common single invasion ancestry.

Tahuna Iti is a stronghold for breeding seabirds, resilient to *R. exulans* which have probably been present for some time (Thibault 1976). Seabirds on Tahuna Iti are jointly threatened by *R. rattus* invasion and human disturbance from eco-tourism operating from Papeete since the late 1980s. The vegetation on the five smaller islets (< 10 ha) has important value as these islands were not heavily planted in coconut trees. Reiono and Tahuna Iti have the highest ecological value for their intact flora and avifauna respectively. Eradicating *R. exulans* from Reiono should allow seabirds to recolonise, creating an 'insurance policy' against seabird disturbance on Tahuna Iti, and mitigating disturbance in other parts of the atoll. The risk of rats reinvading the rat-free Reiono and Tahuna Rahi unit is low given their isolation (1150 m).

Although Tetiaroa appears generally pristine due to uninhabitation, the ecosystem is degraded by introduced species. Introduced rats limit the distribution of seabirds, where Tetiaroa is their last stronghold in the Society Islands. Introduced plant species on Onetahi and Rimatuu are naturally spreading (Sachet and Fosberg 1983, Butaud 2006). New invasions continue, such as a small but growing number of red-vented bulbuls (*Pycnonotus cafer*) observed on Tahuna Iti and Rimatuu in the last few years (Butaud 2006), and a pair of common mynas (*Acridotheres tristis*) observed on Onetahi in January 2010. In both cases colonisation was likely by self-dispersal from Tahiti or Moorea. Eradication of small populations of plants and birds before they become established should be considered a priority management action. Other species are likely arriving unnoticed (e.g., insects). Ongoing biosecurity quarantine and surveillance is required.

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Variable efficacy of rat control in conserving Oahu elepaio populations

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Abstract The Oahu elepaio (*Chasiempis ibidis*), an endangered forest bird endemic to the Hawaiian island of Oahu, is threatened by nest predation from alien ship rats (*Rattus rattus*). Rat control has been implemented in several areas to reduce nest predation, but success of most control programmes has not been assessed previously. We evaluated responses of elepaio to rat control at six sites from 2000-2009; determined cause(s) of poor performance at some sites; and recommended ways to improve elepaio conservation through adaptive management. Rats were controlled during the elepaio nesting season with snap traps and bait stations containing 0.005% diphacinone. Rat control resulted in overall improvements in elepaio fecundity (50%), female survival (10%), and population growth (18%), but efficacy varied among sites, and performance was related to effort. Elepaio continued to decline at some sites despite rat control. Control programmes performed well at three sites where frequency of management and density of bait stations and traps were higher. Performance was compromised at two sites by infrequent or irregular access. At two sites with sparse elepaio populations, efficacy may have been reduced by patchy distribution and low density of bait stations and traps. Increasing the frequency of management and density of traps and bait stations should improve performance, and the scale of rat control should be expanded at all sites to keep pace with elepaio population growth. Alternative management strategies also should be investigated, including broadcast application of rodenticide, use of large-scale trapping grids to create predator-free mainland islands, construction of predator-proof fences, and restoration of native tree species.

Keywords: Hawaii, *Chasiempis ibidis*, ship rat, *Rattus rattus*, predation, productivity

INTRODUCTION

Introduced predators are widely recognised as one of the most serious threats to island species worldwide (Blackburn *et al*. 2004), and the ship rat *Rattus rattus* is perhaps the most pervasive alien predator, particularly of island birds (Jones *et al*. 2008; Drake and Hunt 2009). Predator control is often used as a means of alleviating predation, and though not always effective (Côté and Sutherland 1997), it has been extremely important in conservation of several species of endangered Pacific island birds (Robertson *et al.* 1994; O'Donnell *et al.* 1996; Moorhouse *et al*. 2003). Assessing the effectiveness of predator control, including performance of control methods and response of the desired species is crucial for achieving success and improving conservation efforts through adaptive management (Innes *et al*. 1999; Choquenot and Parkes 2001; Armstrong *et al.* 2006).

The Oahu elepaio (*Chasiempis ibidis*) is a territorial, non-migratory monarch flycatcher endemic to the Hawaiian island of Oahu (VanderWerf 1998). Elepaio also occur on the islands of Kauai and Hawaii, but the forms on each island recently were split into separate species based on morphological, behavioural, and genetic evidence (VanderWerf 2007; VanderWerf *et al*. 2009; Chesser *et al.* 2010). The Kauai elepaio (*C. sclateri*) and Hawaii elepaio (*C. sandwichensis*) are fairly common (Scott *et al*. 1986), but the Oahu elepaio is listed as endangered under the U.S. Endangered Species Act (USFWS 2006) and by the State of Hawaii. The Oahu elepaio has declined rapidly over the past few decades and now occupies only 4% of its presumed prehistoric range (VanderWerf *et al*. 2001). In the 1990s, island-wide population of Oahu elapaio was estimated as about 1,980 birds (VanderWerf *et al*. 2001), but it has declined since then. The distribution of elepaio is highly fragmented, with six relatively large populations estimated at 100 or more birds and numerous small relicts with just a few birds (Fig. 1).

Artificial nest experiments with remote cameras have revealed high predation rates in Oahu elepaio habitat, with ship rats as the most common nest predator (VanderWerf 2001). Some elepaio also die from avian poxvirus (*Poxvirus avium*) and probably from avian malaria (*Plasmodium relictum*; VanderWerf *et al*. 2006), but nest predation is a more serious threat (VanderWerf 2009). Elepaio are mostly confined to areas protected from development, but degradation of forest habitat by invasive alien plants and feral ungulates is an ongoing threat in much of their range. There may be occasional predation on adult elepaio by feral cats (*Felis catus*). Because recently fledged elepaio sometimes leave the nest before they can fly well and spend time on or near the ground (VanderWerf 1998), they are vulnerable to predators such as feral cats, small Indian mongoose (*Herpestes auropunctatus*), and feral pigs (*Sus scrofa*).

A rat control programme using snap traps and bait stations with diphacinone, which began in 1996 in the SE Koolau Mountains in an effort to stop elepaio population declines, has proved to be an effective means of increasing nest success and survival of breeding females (VanderWerf and Smith 2002; VanderWerf 2009). Based on this

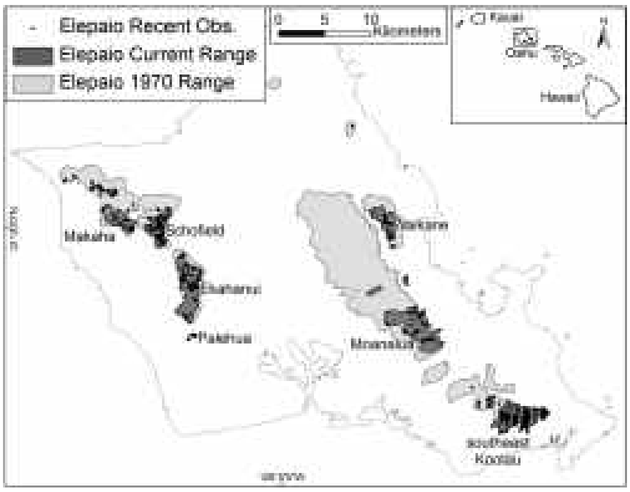

Fig. 1 Oahu elepaio distribution and study site locations.

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success, rat control has been implemented in several areas on Oahu by multiple agencies and organisations (USFWS 2006; VanderWerf 2007; U.S. Army 2009). However, the success of most control programmes has not been assessed previously. In this paper we: 1) evaluate response of elepaio to rat control programmes at six sites and compare efficacy among sites; 2) determine cause(s) of poor performance at some sites; and 3) make recommendations for improving elepaio conservation efforts through adaptive management.

METHODS

Study Sites

Rat control and elepaio monitoring have been conducted at seven sites on Oahu that encompass different portions of the species' current range (Fig. 1). Work began at the SE Koolau site in 1996, followed by addition of Schofield Barracks and Ekahanui in 2000, Makaha in 2005, Moanalua in 2006, and Palehua and Waikane in 2007. Sizes of the study sites ranged from 32-117 ha, and density of elepaio also varied (Table 1). Some sites contained many elepaio territories that were closely spaced (southeast Koolau, Ekahanui, Palehua, Moanalua), but in others elepaio were more sparsely distributed, with gaps between territories (Makaha, Waikane). The SE Koolau and Schofield sites consisted of multiple sub-sites comprising adjacent valleys. Habitat in the study sites was wet or mesic forest, with average annual rainfall ranging from 980 mm at Palehua to 3750 mm at Waikane. Average elevation ranged from 180 m in the SE Koolau to 730 m at Schofield. All sites were dominated by alien plants, particularly strawberry guava (*Psidium cattleianum*), christmasberry (*Schinus terebinthifolius*), and kukui (*Aleurites moluccana*), but some sites, particularly Ekahanui and Moanalua, also contained a substantial amount of native vegetation. In some cases a portion of each site was monitored but not managed due to staffing constraints and was used as a control in which elepaio demographic rates were measured in the absence of rat removal. Because the SE Koolau site has been monitored longest and managed most consistently (VanderWerf 2009), information from that site was used as a benchmark for comparison with other sites.

Rat Control

Rats were controlled at each site using a combination of trapping and toxicants. Eaton's bait blocks (J.T. Eaton Inc., Twinsburg, Ohio, USA) or Ramik mini-bars (HACCO Inc., Randolph, Wisconsin, USA) containing 0.005% diphacinone were placed in tamper-resistant Protecta plastic bait stations (Bell Laboratories, Madison, Wisconsin, USA) to shield them from rain and to reduce the risk of poisoning non-target species. Bait stations were secured in trees at least one metre off the ground to restrict access by dogs (*Canis familiaris*) and feral pigs. During each check, up to 454 g (16 oz) of bait were added to each station and any spoiled bait was removed. Application of diphacinone bait was conducted in compliance with U.S. Environmental Protection Agency registration numbers 61282-26 and 56-42 and special local need registrations HI-940001, HI-960005, and HI-980008. Victor Professional rat snap traps (Woodstream Corp., Lititz, Pennsylvania, USA) were used to augment the control, allow identification of rat species present, and provide a measure of relative rat abundance. Traps were tied to trees or rocks to prevent scavengers from removing them but were not covered. Traps were counted as having caught a rodent if hair or tissue was stuck to the trap, and traps were cleaned with a wire brush after each capture to remove evidence of previous captures.

Rat control commenced in late December or early January each year, about one month before the elepaio nesting season, and ended after the last known nest either fledged chicks or failed, usually in late May or June. From two to four bait stations and two to four snap traps were deployed in elepaio territories known to contain a breeding pair, and sometimes in territories of single males, but not in gaps between territories. Traps and bait stations were deliberately concentrated in sections of each territory known to have been used habitually for nesting, if such information was available. Elepaio territory size varies with habitat structure (VanderWerf 2004) and ranged from 1.0-2.0 ha among sites. Variation in elepaio population density and territory size lead to unintended variation in density of traps and bait stations among sites (Table 1). Traps and bait stations were checked and rebaited weekly in most cases, but in some areas the frequency of maintenance was lower (Table 1). At Schofield Barracks, access sometimes was restricted by military training, resulting in either less frequent maintenance or maintenance of only portions of the study area. Waikane was also visited less consistently, due to staffing limitations and difficulties in accessing the site during wet weather. The effect of effort on performance of rat control was investigated with a multiple regression analysis using number of rats caught per trap per visit as the dependent variable, and three measures of effort (number of visits, density of traps, and density of bait stations), as independent variables.

Elepaio monitoring

Elepaio were monitored on weekly visits to each territory during the nesting season, usually in conjunction with maintenance of traps and bait stations, and occasionally in other months outside the nesting season. Elepaio territories were identified using song playbacks and spot-mapping (VanderWerf *et al*. 2001; VanderWerf 2004). Some elepaio at each site were captured with mist-nets and marked with a metal leg band and a unique combination of three plastic coloured leg bands to facilitate monitoring. A total of 152 elepaio were banded at all sites combined, including 124 males and 28 females. The sample of females was smaller because they responded less aggressively to playbacks and

Table 1 Summary of rat control effort and performance by site. Number of bait stations and traps are the maximum used in any year at that site.

Study Site	Size (ha)	Max. Territories managed	Total # bait stations/ traps	Density of bait stations/ traps /ha	Visits per year	Average bait take	Average rats/ trap/visit
Palehua	32.9	19	37/37	1.1/1.1	13.3	22%	0.12
Moanalua	117.1	29	87/174	0.7/1.5	16.5	11%	0.12
Ekahanui	31.9	27	68/124	2.1/3.9	16.8	17%	0.07
SE Koolau	73.2	47	71/79	1.0/1.1	15.4	17%	0.11
Schofield	64.2	24	95/178	1.5/2.8	4.6	46%	0.22
Waikane	39.6	7	32/64	0.8/1.6	5.5	38%	0.27
Makaha	88.6	13	39/72	0.4/0.8	14	18%	0.19

were more difficult to capture. Nests were searched for and monitored, and counted as successful if they fledged at least one chick. Elepaio fecundity was measured as the number of fledglings produced per pair each year.

Annual survival of adult elepaio was estimated using multi-state mark-recapture models in program MARK, with birds grouped by sex, and separate states for rat control and no rat control. Sample sizes were too small to estimate survival at each site individually, but sites were divided into two groups based on whether elepaio numbers were stable or increasing (Ekahanui, Moanalua, Palehua) or declining (Schofield Barracks, Makaha, Waikane). Juvenile survival was estimated by enumeration, which is simply the proportion of surviving birds, because few juvenile elepaio have been captured on Oahu (n = 6). VanderWerf (2009) provides more detail on use of mark-recapture models to estimate elepaio survival.

The finite rate of elepaio population growth, or lambda, was calculated for each site using a simple formula from Pulliam (1988): λ= Adult survival + (fecundity x juvenile survival). Values of lambda > 1.0 indicate population increase, those < 1.0 indicate decline, and a value not different from 1.0 indicates no change. Annual survival of females was used for adult survival because it was lower than survival of males and thus limited population growth (Kilpatrick 2006). All values are reported as mean ± SE unless otherwise noted.

RESULTS

Rat Control

Performance of rat control varied among sites, and this was due, at least in part, to variation in effort (Table 1). At sites where bait stations and snap traps were maintained

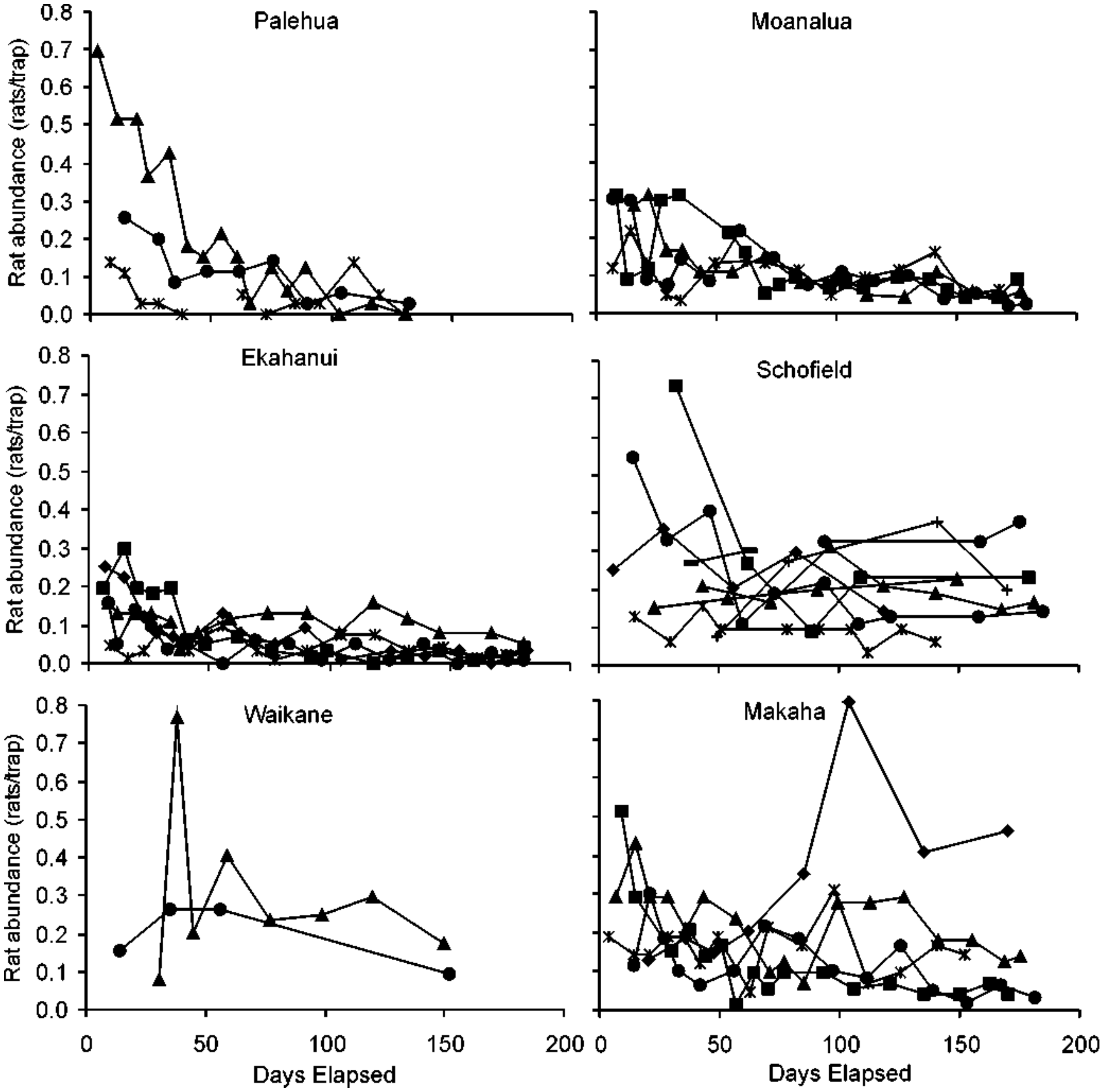

Fig. 2 Rat trapping rate over time at each study site. Each line represents a different year. Trapping rate failed to decline at Schofield, Waikane, and Makaha due to infrequent management or lower density of bait stations and traps.

more frequently (Palehua, Moanalua, Ekahanui, SE Koolau), rat abundance declined after 30-40 days and usually remained low thereafter (Fig. 2). In contrast, at sites that were maintained less often or irregularly (Schofield, Waikane, Makaha), rat abundance fluctuated over time and often failed to reach the low levels observed at other sites. Similarly, average bait take and rat trapping rate were lower over the entire season at sites with regular management (Fig. 3). Rat abundance and performance of rat control also varied among years at most sites, even at sites where control effort was consistent among years (Fig. 3). Only at Moanalua was rat abundance consistently low each year. Multiple regression confirmed that rat

Table 2 Regression of rat control effort measures on performance. Number of visits was most closely related to performance.

Measure of Effort	T	p-value
Number of visits	-5.22	<0.001
Bait station density	-1.84	0.08
Trap density	0.76	0.45

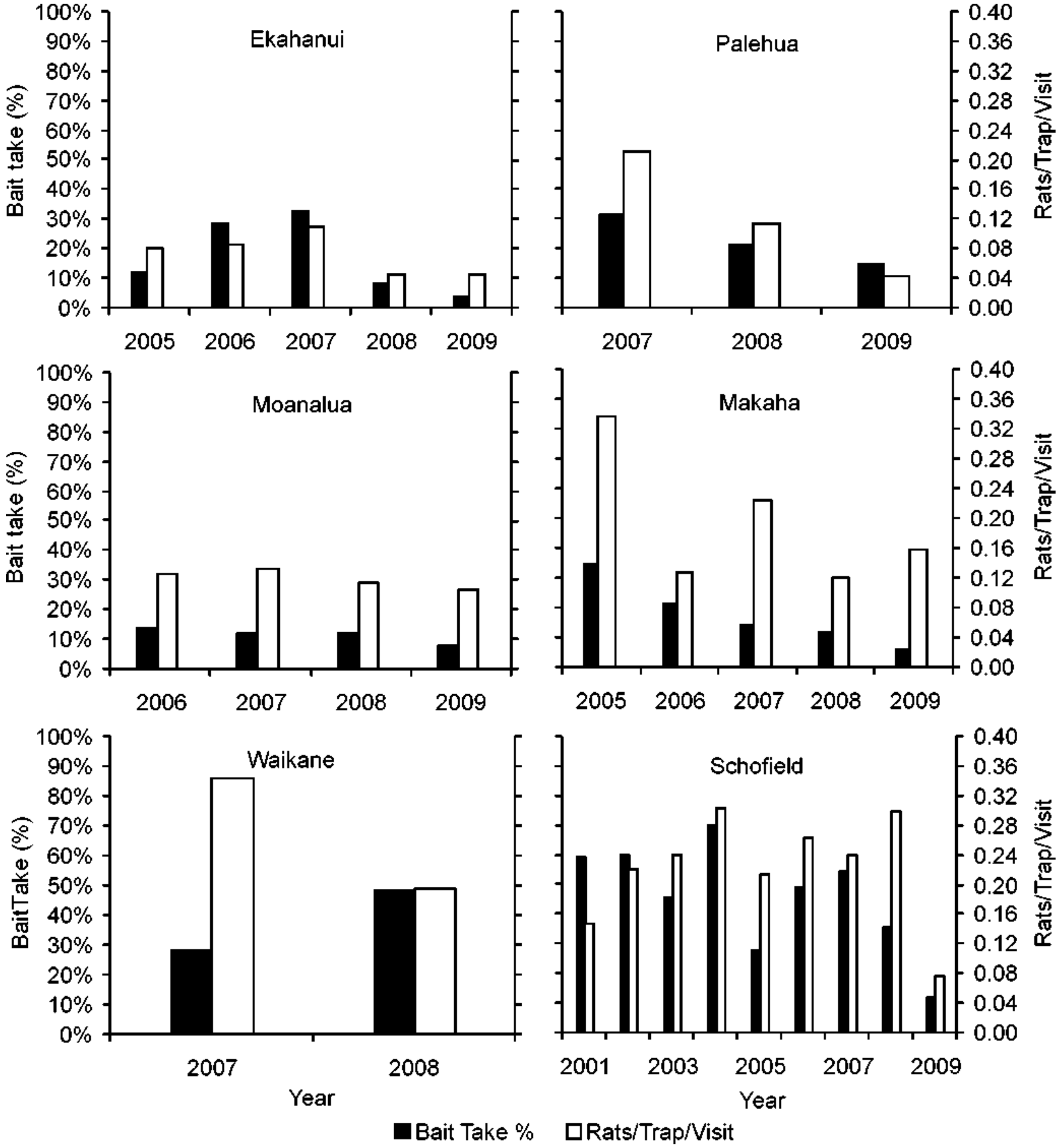

Fig. 3 Annual variation in bait take and rat trapping rate at each study site. Values for each site are averages over the entire season. Bait take and trapping rate were lower at sites where density of traps and bait stations were higher and that were maintained more often.

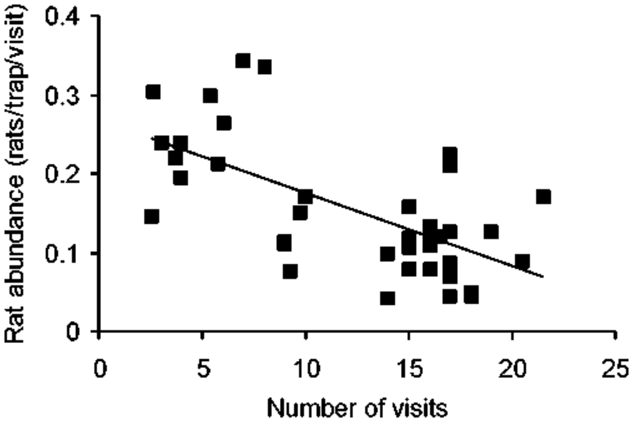

Fig. 4 Relation of management frequency to performance. Each point represents a year at one site.

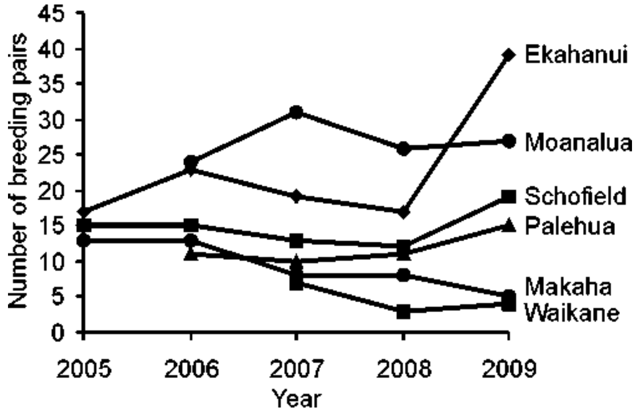

Fig. 7 Oahu elepaio breeding pair numbers at six study sites over time.

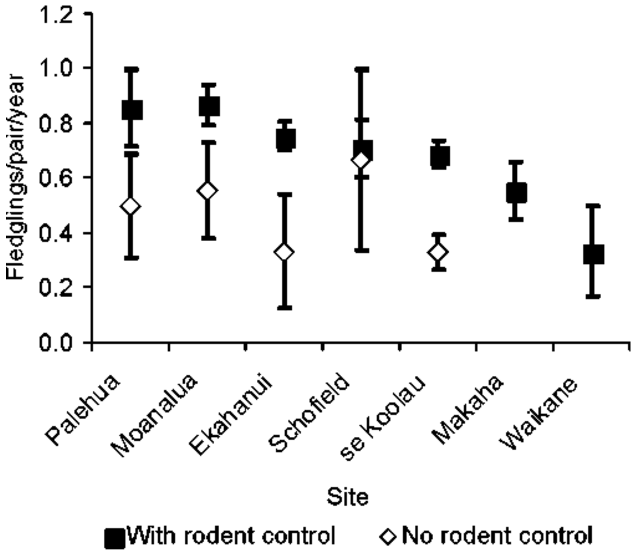

Fig. 5 Productivity of Oahu elepaio at each study site with and without rat control.

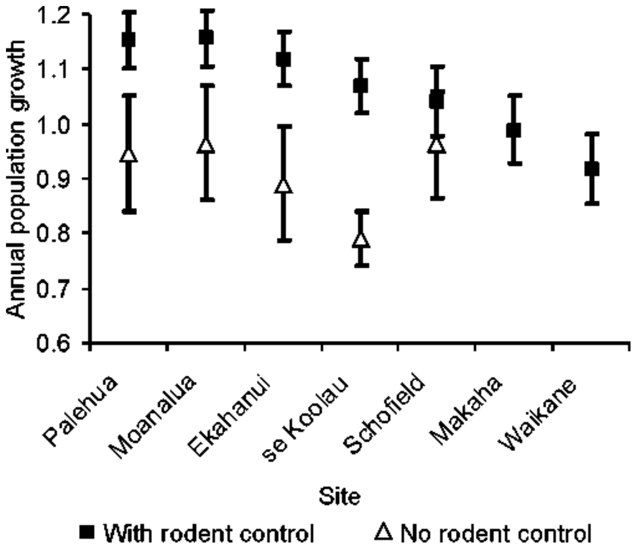

Fig. 6 Oahu elepaio population growth at each site with and without rat control. Values > 1.0 indicate potential growth, values < 1.0 indicate decline.

abundance was related to control effort ($F_{3,34} = 11.13$, $p < 0.001$, $R^2 = 45.1\%$), and further revealed that the number of visits on which traps and bait stations were maintained was most important in reducing rat numbers (Table 2; Fig. 4). Density of bait stations had a weaker relationship with rat abundance, and, surprisingly, trap density was not related to rat abundance.

Elepaio monitoring

Productivity of Oahu elepaio was about 50% higher with rat control (0.75 ± 0.04 fledglings per pair per year) than without (0.52 ± 0.11; $F_{1,408} = 4.04$, $p = 0.04$), in all sites and years combined. However, productivity was lower at some sites, and rat control was less effective at some sites (Fig. 5).

Annual survival of adult female Oahu elepaio was 10% higher with rat control (0.84 ± 0.05) than without (0.74 ± 0.09). Survival of males was less affected by rat control (0.88 with ± 0.02 vs. 0.85 ± 0.03 without), presumably because only females attend the nest at night, when rats are most active. Annual survival of adults was 3-6% higher at sites where elepaio numbers were stable or increasing (Ekahanui, Palehua, Moanalua) than at sites where elepaio were declining (Schofield, Makaha, Waikane). This difference in survival was evident with (5-6%) and without (3-4%) rat control, indicating there was some difference among sites that was not corrected by rat control. Annual survival of juvenile elepaio was 0.33, but this estimate was based on a very small sample.

Elepaio population growth was about 18% higher overall with rat control (1.09 ± 0.05) than without (0.91 ± 0.09), but performance varied among sites and growth was not positive in all cases (Fig. 6). Observed trends in elepaio numbers (Fig. 7) generally agreed with demographic calculations; elepaio numbers grew at sites where lambda was > 1.0 and declined at sites where lambda was < 1.0.

DISCUSSION

Rat control was generally effective at reducing predation on Oahu elepaio nests. Overall rates of elepaio productivity (0.75 ± 0.04), female survival (0.84±0.05), and population growth (1.09 ± 0.05) were substantially improved by rat control, and were similar to those found in a longer-term study in the SE Koolau Mountains (0.69 ± 0.05, 0.82 ± 0.05, and 1.07 ± 0.04, respectively; VanderWerf 2009). Rat control continues to be an effective management tool for Oahu elepaio and is the cornerstone of the recovery strategy for this species.

However, rat control did not perform equally well at all sites, and at some sites elepaio numbers declined despite rat control. Performance of rat control was affected by multiple factors, including frequency of management, density of bait stations, relative size of the managed area, and prevalence of other threats such as disease. Management at some sites should be continued using current methods, but at other sites improvements are needed and alternative management strategies should be investigated.

At Palehua, Ekahanui, and Moanalua, rat control performed well and elepaio numbers grew in response. No changes are necessary to methods used at these sites at this time, except that the scale of rat control programmes must expand to keep pace with elepaio population growth to allow continued recovery. The area managed at these sites included most or all of the elepaio in the area, and this contributed to their success. At the SE Koolau site, rat control performed well, but elepaio numbers continued to decline despite a lambda value > 1.0 (VanderWerf 2009). This apparent paradox arose because the SE Koolau study site encompassed only a fraction of the largest remaining elepaio population on the island (VanderWerf *et al*. 2001), allowing some young birds to disperse into adjacent unmanaged areas that acted as sinks. The study site served as a "pseudo-source" from which elepaio emigrated even though there was little surplus. Rat control must be expanded at this site to reduce the edge effect or elepaio numbers will continue to decline until source-sink equilibrium is reached. At Palehua, Ekahanui, and Moanalua, this situation can be avoided if rat control is expanded as necessary each year so it continues to encompass most of the growing elepaio population.

Effectiveness of rat control was compromised by lower effort at Schofield, Waikane, and Makaha. Inadequate frequency of management was the most serious limitation, but low density bait stations also may have contributed to poor performance. Diphacinone is a first-generation anticoagulant, and rats must consume bait for several consecutive days in order to ingest a lethal dose. If frequency of management is not sufficient to ensure an uninterrupted supply of bait or if the distance between stations is larger than rat home range size, then rat control will be difficult. A minimum of 10 visits appeared necessary to achieve effective control using diphacinone bait stations (Fig. 4), but this may vary among sites and years depending on rat abundance. In Hawaii, label requirements for diphacinone use for conservation purposes specify that bait stations should be spaced at an interval of 25 to 50 metres, and this was adhered to within territories at all sites, but the overall density of bait stations was lower at Waikane and Makaha because there were large gaps between some territories. Density of snap traps was less important than density of bait stations, suggesting bait stations played a larger role in controlling rodent numbers, at least over the range of trap densities used in this study.

At Schofield Barracks, although the density of bait stations and traps was high, frequency of maintenance was low and irregular due to access restrictions imposed by military training, and this compromised efficacy of the rat control programme in some years. Frequency of access was highest in 2009, when rat control was also most effective (Fig. 3). The effect of rat control appeared to be lower at Schofield than at other sites, but this may have been an artefact of the compromised control programme. Elepaio territories were categorised each year as either having or not having rat control, but in reality the distinction between these treatments at Schofield was less clear because irregular trapping and bait station maintenance only led to partial suppression of rat abundance. Access to Schofield is unlikely to improve in the long-term, so achieving effective control with diphacinone bait stations may continue to be problematic, and pursuit of an alternative approach is warranted. Coincidentally, a two-year window is available from 2010-2011 during which more regular access will be possible, and plans are underway to construct an ungulate fence around a 1000 ha area encompassing most of the elepaio population at Schofield Barracks, as well as 16 endangered plant species and multiple small populations of the endangered tree snail *Achatinella mustelina.* Once feral pigs are removed from the fenced area, aerial or hand broadcast of rodenticide may be possible, which would require fewer visits to achieve effective rat control and potentially could protect a larger portion of the population.

The rat control programme in Waikane also suffered from infrequent and irregular maintenance, but this was due to difficultly in accessing the site via a rough road during wet weather, and occasionally to staffing limitations. Steep terrain and deep ravines also limited placement of bait stations and traps, and made it difficult to place them in proximity to nest trees, and the location of some nests was unknown. More frequent maintenance and higher density of bait stations may have improved results, but the low number of elepaio at this site made it less cost-effective to manage, and it was discontinued in 2009.

At Makaha there were no restrictions on access and bait stations and traps were maintained frequently, and the number of stations and traps deployed in each elepaio territory was similar to other sites. However, because elepaio at Makaha were sparsely distributed with large gaps between breeding pairs, bait stations and traps were less uniformly distributed and their density was the lowest of any site. Bait take and trapping rates were high but failed to decline, probably due to reinvasion by rats from intervening gaps. Prevalence of avian poxvirus was particularly high in Makaha and nearby areas (VanderWerf *et al*. 2006), and it is possible increased mortality from disease counteracted any improvement achieved through predator control. Deploying bait stations and traps in a more uniform pattern over the whole valley might improve performance, but would be less cost-effective because so few elepaio remain (three pairs). Management of this site was discontinued in 2010 in order to focus efforts on other areas.

Although rat control has been effective, alternative management techniques are worth investigating in order to provide a more comprehensive conservation strategy. Most Oahu elepaio nest in alien trees that bear fruit or nuts attractive to rats, not because elepaio prefer these plant species, but rather because they are the dominant plants in areas where elepaio remain (VanderWerf 2009). Restoration of native trees that are less attractive to rats would benefit elepaio by providing safer nest sites and may be a means of reducing the need for rat control. If alien trees are removed, simultaneous reforestation with native species would minimise any disruption of nest site availability and foraging habitat. In order to achieve meaningful recovery at a landscape scale, predation must be managed over larger areas. This has been achieved in several areas of New Zealand through construction of predator-proof fences and permanent eradication of rats and other predators, and use of large predator control grids to create predator-free "mainland islands" (Clout 2001; Dilks *et al*. 2003; Parkes and Murphy 2003; Saunders and Norton 2001).

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Attempting to eradicate invasive Gambian giant pouched rats (*Cricetomys gambianus*) in the United States: lessons learned

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Abstract Gambian giant pouched rats (*Cricetomys gambianus*) are native to Africa, but they are popular pets in the United States. They caused a monkeypox outbreak in the Midwestern United States in 2003 in which 72 people were infected. A free-ranging population became established on the 400 ha Grassy Key in the Florida Keys, apparently after a release by a pet breeder. This rodent species is known to cause extensive crop damage in Africa and if it reaches the mainland US, many impacts, especially to the agriculture industry of Florida, can be expected. An apparently successful inter-agency eradication effort has run for just over three years. We discuss the strategy that has been employed and some of the difficulties encountered, especially our inability to ensure that every animal could be put at risk, which is one of the prime pre-requisites for successful eradication. We also discuss some of the recent research with rodenticides and attractants, using captive Gambian rats, that may help with future control and eradication efforts.

Keywords: Bait station, Florida, inter-agency project, rodent, traps, zinc phosphide, human attitudes

INTRODUCTION

Introduced omnivorous rodents have endangered or eradicated numerous native species on islands where the rodents have few or no predators (Moors and Atkinson 1984; Veitch and Clout 2002; Engeman *et al*. 2006; Witmer *et al*. 1998). For example, most seabirds that nest on islands have not evolved to deal with mammalian predation and are very vulnerable to introduced rodents and other species introductions. In response, there has been a concerted worldwide effort to eradicate introduced rodents from uninhabited islands, often successfully (Howald *et al*. 2007). These efforts have relied heavily on the use of rodenticides (Howald *et al*. 2007; Witmer *et al*. 2007a). While eradication is generally the preferred management approach to an invasive vertebrate species (e.g., Panzacchi *et al*. 2007), in some situations, sustained control is the only viable option (Parkes 1993; Parkes and Murphy 2003).

Native to Africa, Gambian giant pouched rats or Gambian rats (*Cricetomys gambianus*) are an invasive species on the island of Grassy Key, Florida (Engeman *et al*. 2006). Gambian rats shifted from a domestic pet to invading species after a suspected release by a pet breeder (Perry *et al*. 2006). Because of their large size (i.e., up to 1 m in length and 2.8 kg in mass; Kingdon 1974), Gambian rats pose a serious threat to native species (e.g., particularly nesting species) and agricultural crops (Fiedler 1998), especially if they rats invade mainland Florida where there is intensive agriculture (Peterson *et al*. 2006). Gambian rats also transmit disease and in 2003 were implicated as facilitators of a monkeypox outbreak that infected 72 people in the Midwestern United States (Enserink 2003).

In this paper, we describe an attempt to eradicate Gambian rats from the Florida Keys, USA. The United States Department of Agriculture, Wildlife Services (WS) initiated eradication and detection efforts in the Florida Keys, but trapping the sparse population of Gambian rats after a rodenticide baiting operation required a lengthy period of time. Trapping is commonly used as part of eradication efforts for carnivores (e.g., Bloomer and Bester 1992, Ebbert 2000, Nogales *et al*. 2003) and feral ungulates (Campbell and Donlan 2005; Lowney *et al*. 2005), but rarely for small rodents. However, long-term trapping efforts have successfully removed some large-bodied, invasive rodent populations including nutria (*Myocastor coypus*) and muskrats (*Ondatra zibethicus*) in the United Kingdom (Gosling and Baker 1989) and nutria at the Blackwater National Wildlife Refuge in Maryland USA (Kendrot and Sullivan 2009). Other efforts to eliminate invasive rodents with trapping have been less successful (e.g., Carter and Leonard 2002; Panzacchi *et al*. 2007). The effort on Grassy Key has been a collaboration of WS, Florida Wildlife Commission (FWC), Florida Parks, United States Fish and Wildlife Service (FWS), and the South Florida Water Management District (SFWMD) and was designed to copy the successful eradication of ship rats (*Rattus rattus*) from Buck Island in the U.S. Virgin Islands (Witmer *et al*. 2007a).

ERADICATION AREA

Grassy Key is a part of the Florida Keys, which extend from the southern tip of Florida and curve south and westward into the Gulf of Mexico. Most of the islands are connected by the major highway, U.S. Highway 1, so the islands are not truly isolated. Grassy Key is about 400 hectares and of very low relief ($\leq$ 2 m above mean sea level). The substrate is coral and the water table is very near the surface so that there is often standing water in some areas. The vegetation consists of a mixture of native and invasive species (Long and Lakela 1971; FNAI 1990) including various species of mangroves, palms, Australian pine (*Casuarina equisetifolia*), Brazilian Pepper (*Schinus terebinthifolius*), and numerous ornamental plant species. Periodic tropical storms and hurricanes damage vegetation and structures, and flood many areas. There are about 300 private residential properties on the island, the majority of which are $\leq$ 1 ha in size. In total, these properties comprise about 40% of the island area.

METHODS

In 2006-07, WS conducted Gambian rat distribution surveys on Grassy Key, using cage traps and motion-sensitive cameras. Gambian rats were found over much of the island with the exception of some areas of standing water. Surveys on other islands of the Florida Keys did not reveal any Gambian rats. Two animals were radio-collared and monitored for about a week, during which time they ranged at least 60 m per day. The survey and movement data served as the basis for the spacing of a bait station grid over the entire island. In the "core area" (residential areas known to support relatively large numbers of Gambian rats), we used a 40 by 40 m grid spacing, whereas, in other areas, we used a 50 by 50 m grid spacing (Fig. 1). The

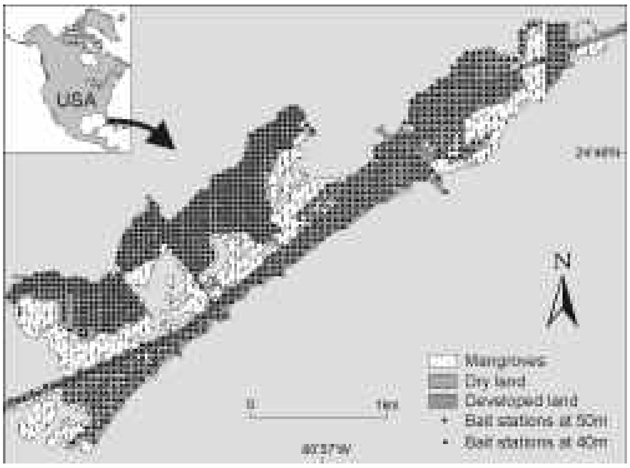

Fig. 1 The grid of bait stations used in the Gambian giant pouched rat eradication attempt, Grassy Key, Florida. US Highway 1 runs the length of the island.

SFWMD hired private contractors to cut trails through dense vegetation in order to establish the grid and provide access to bait stations. GPS units were used to assist with the establishment of a symmetrical, consistently spaced grid of approximately 1000 bait stations over the 400 ha. Six private properties, totalling about 2 ha in area, did not allow access by WS personnel.

WS conducted preliminary rodenticide bait trials, using wild-caught animals maintained in pens, with a variety of commercial baits, including several anticoagulants and a zinc phosphide (ZP)-grain mix. The ZP bait seemed the most efficacious, resulting in 100% mortality in a short period of time (generally a few hours or less) after consumption of a few grams of the bait in a single feeding session. The final bait formulation consisted of mostly peanut butter with some horse sweet mix (mainly grains and molasses), and enough ZP concentrate to result in an active ingredient concentration of 2%. This mixture formed a paste that could not be readily removed from the bait stations, thus reducing the risk of non-target animal exposure to the bait. WS also designed a bait station that allowed access by Gambian rats, but seemed to prevent access by most non-target raccoons (*Procyon lotor*) , opossums (*Didelphis virginiana*), cats (*Felis catus*) and dogs (*Canis familiaris*), based on remote camera surveillance (Fig. 2).

Fig. 2 Bait station designed and used in the Gambian giant pouched rat eradication attempt on Grassy Key, Florida.

The large number of bait stations relative to staff available precluded filling and monitoring of all bait stations in less that several days. Hence, WS used a "rolling front" strategy whereby the island was divided longitudinally into zones. Bait was applied to one zone at a time, moving from east to west. The operation started with a 3-day pre-baiting period in which grain mixed with peanut butter was placed in the bait stations to get Gambian rats used to entering the bait stations for food. Next, ZP bait was placed and maintained in the stations during late May and early June 2007.

Before, during, and after the baiting session, cage traps and remote cameras were also used to detect and remove individual Gambian rats. If a Gambian rat was detected by one of the cameras, several cage traps were set in the area and nearby bait stations were filled with the ZP bait. Captured rats were euthanased by gunshot to the head. When non-target animals (raccoons, opossums) were captured in a cage trap, they were released on a nearby island as directed by the FWC. This reduced non-target mortalities and cage trap interference which was reducing the efficacy of trapping the target species. Any ship rats, another invasive rodent in Florida, captured were euthanased.

An additional baiting session was conducted in September 2007, in the same manner as previously described along with intensive trapping in those areas still inhabited by Gambian rats. Additionally, a different formulation of the ZP bait was used (no peanut butter, but with cantaloupe oil added) and WS switched from baiting cage traps with peanut butter to cantaloupe fruit. These changes were made because it was believed that the remaining rats might not be attracted to the previous baits used in bait stations and cage traps.

For many species of rodents, an eradication can be considered successful if intensive, periodic surveys do not reveal any individuals of the target species for two years (Witmer *et al*. 2007b). This did not happen in the first 2.5 years after the initial eradication effort, despite 280 cage traps and 80 remote cameras being used in the subsequent "mop-up" effort.

RESULTS

Within a few days, the field crew could smell decomposing carcasses in some areas, even though no carcasses were found on the surface during field work. However, camera surveillance soon made it clear that some Gambian giant pouched rats remained after the main baiting effort in May-June 2007.

Captures of Gambian rats steadily declined from September 2007-2009. Between May and August 2008 only 19 Gambian rats were caught. A hurricane before this period may also have killed numerous individuals. After several months of no captures, an adult female Gambian rat was captured in September 2009. She was radio-collared and found to rarely leave a 1 ha private property that WS was not permitted access to during the eradication programme. Of the six private properties that WS did not have access to, five were $\leq$ 0.2 ha and one, of about 1 ha, was where the last Gambian rat was caught and radio-collared. Intensive trapping was conducted around these properties throughout the eradication effort. While these areas were only about 2 ha of the 400 ha island, they may be an important contributor to the protracted eradication effort. We believe that the radio-collared female is now dead as her radio-signal location has not changed from a limestone structure on the property for over 6 months. An intensive two-week trapping and camera session in June 2010 using 300 cage traps and about 40 remote cameras did not reveal the presence of any Gambian rats. WS is working with the FWC to establish a quarterly monitoring schedule for the next two years.

Evidence of the potential for emigration from Grassy Key towards mainland Florida emerged during the eradication. In 2008, a single, dead (presumably vehicle-killed) Gambian rat was reported along a highway in Islamorada, on Upper Matecumbe Key. WS confirmed that the dead animal was a Gambian rat. This Key is about 33 km east of Grassy Key and about half way to the mainland of Florida from Grassy Key. The Key is linked to Grassy Key by multiple bridges, some of which are several kilometres long. Cage traps and motion-sensitive cameras were set in a grid in the area and operated for several days after the carcass was discovered. No further Gambian rats have been detected on Upper Matecumbe Key and its origins remain unclear. This example illustrates the need for a good bio-security system if we are to prevent invasions by foreign species and their spread from infested areas (Broome 2007).

Additional research has been conducted with wild-caught Gambian rats from Grassy Key at the WS' National Wildlife Research Center in Fort Collins, Colorado, and has identified other potential attractants (Witmer *et al*. 2010a) and rodenticides (Witmer *et al*. 2010b) for use in future efforts with invasive Gambian rats wherever they may show up. Hopefully, the invasive rodent eradication effort on Grassy Key will end with the complete removal of all Gambian giant pouched rats, if any still remain on the island.

DISCUSSION

Recent intensive trapping and camera monitoring suggests that eradication has been achieved, but it will take additional monitoring to verify success. We found that, despite extensive eradication and detection efforts by WS in the Florida Keys, detecting and trapping the presumably few remaining Gambian rats on Grassy Key proved difficult. We know that getting the last few individuals in an eradication effort is often the most difficult part of the project and is virtually impossible if there are refuges available that protect some individuals from the eradication technology. Hence, a 99% success in an eradication attempt generally means the operation has failed. Some of the following factors may have contributed to the protracted effort Grassy Key.

Lack of data on the target species. Most rodent eradications deal with species of *Rattus* and *Mus*. Compared with these, relatively little was known about the biology and ecology of the Gambian rats on Grassy Key before we started the eradication project. While a rapid response to a newly discovered invasion is necessary for achieving a successful eradication before wide dispersal and establishment, it is important to understand the species and its use of its new environment. Published literature on Gambian rats is sparse and unpublished and/or obscure sources in Africa are not readily available to us in the United States except for informative websites maintained by persons keeping exotic pets. Time and funds permitting, the Gambian rats on Grassy Key should have been more intensely studied before the eradication effort. If Gambian rats ultimately survive this eradication effort, aspects of their behavioural ecology should be studied that will enable better design of an eradication strategy.

Adequate funding and resources are essential to successful invasive species eradication. We faced funding and staffing limitations from the start. We often worked on a "shoe string" budget which made planning and execution of the project difficult at best. There were times when funds and field staff were not available for a period of time during the eradication. At times, we functioned with one person in the field. Efficient planning and use of funds and staff help with these conditions, but cannot totally overcome the problem. Eradications require contingency planning and quick actions after unexpected occurrences or situations — these responses require adequate funds at hand.

Public cooperation and universal land access for operators are crucial to an invasive species eradication effort. Meeting with landowners is very important to help gain their trust and cooperation. Taking a list of predetermined talking points to public meetings can be very useful because proposed residential eradication attempts will draw much attention from the public and media. In the case of Grassy Key, most property was privately owned. While most landowners cooperated with the eradication effort and allowed access to their property, some did not, thereby causing a violation of the most important pre-requisite for successful eradication: that there be no refuges where individuals can avoid detection and removal. The last remaining Gambian rats seem to be associated with the six inaccessible properties. Based on limited radio-telemetry data, it appears that those Gambian rats found all they needed (food, water, shelter) on a single property and rarely left it. Because these few properties were small in size (< 1 ha), our recourse was to place cage traps (and in some cases, bait stations) around the perimeter of those properties with the hope that we would remove all the Gambian rats over time. Needless to say, this required a focused effort by our limited staff to check traps, process animals and re-set traps each day over an extended period.

Some property owners support invasive rodent rat eradication, but do not want rodenticide (i.e., toxicants) used on their property. Understandably, there is a general distrust of the use of chemicals in the environment by some individuals which hindered our effort in a few cases. In these situations, as with property owners refusing access to their properties, we had to use labour-intensive cage trapping over an extended period of time.

Human attitudes often cause unexpected problems for invasive species control in inhabited areas. On Grassy Key, some local residents maintained feeding and watering stations for feral cats. These resources might unintentionally support Gambian rats and other invasive species. Some people will also spring cage traps, damage or remove traps, or let captured animals loose. In our operation, over 100 cage traps were stolen or destroyed. As well as the waste of WS funds and effort, once an animal has been in a trap and then turned loose, it may become trap-shy and difficult to capture in future attempts. All these activities can reduce the chances that eradication will succeed.

Severe weather (e.g., tropical storms) on tropical islands is often unpredictable and can hinder eradication efforts. On Grassy Key, Hurricane Katrina damaged vegetation and transect access, disrupted cages, and caused a power outage during part of the eradication operation. Meeting such a challenge requires contingency planning activities and extra resource commitment, and prolongs the eradication project and increases its cost. On the other hand, it is often important to incorporate seasonal weather conditions into the eradication process to take advantage of, for example, periods when migratory birds are not present or when natural food resources for rodents are scarce so that the rodents will be attracted to rodenticide baits or baited traps.

When there is an unexpected leap or dispersal event of the localised invasive species during an eradication, resources have to be diverted to investigate it. This happened when a dead Gambian rat was discovered miles and islands away from Grassy Key. WS sent staff from the Grassy Key operation to investigate the incident. Several days were spent setting up remote cameras and cage traps. No other Gambian rats were detected or captured and the effort was ended with staff returned to resume the eradication effort on Grassy Key.

While this is not meant to be a complete list of complications that arose during our eradication effort, it might remind operators and others of some common difficulties. Finally, while those involved in eradication efforts should be positive in their efforts, they should not prematurely assume or voice a positive outcome before it is achieved. Detection and "mop-up" of the last individuals after an eradication effort can be the most difficult part of the entire operation. Eradications of an established invasive species are difficult at best and not to be undertaken by the weak of heart!

CONCLUDING COMMENTS

Invasive vertebrates are a serious threat to human resources, health and the environment. Efforts to prevent introductions, control, or eradicate these invasive species are warranted and should continue. However, Parkes (1993) noted that "management that is not inclusive of pests, resources, people, and their interactions usually fails." Good collaboration between federal, state, and local governments is essential, as is consultation with stakeholders to ensure the support and cooperation of landowners and to minimise sabotage of the project. Increased public education should help prevent future introductions and encourage rapid reporting, resulting in early response to the invasion. Increased funding (based on risks, hazards, and priorities) is essential to combat the threat of invasive species in the United States and worldwide.

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Eradication of invasive rodents on islands of the United States

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Abstract Many invasive rodents have become established in the United States and its territories. The species include several species of *Rattus*, house mice (*Mus musculus*), Gambian giant pouched rats (*Cricetomys gambianus*), ground squirrels (*Spermophilus parryii*), nutria (*Myocastor coypus*) and marmots (*Marmota caligata*). These rodents have caused serious impacts to native flora and fauna, agriculture, and other resources. Since the early 1990s, agencies have been eradicating rodents from various islands, primarily for conservation purposes. Of about 40 eradication attempts, 22 (55%) appear to have succeeded. For several islands, however, it is too early to determine if the attempted eradication has been successful or not. In the case of failed eradications, rapid re-invasion by rodents from nearby islands may be the reason. Numerous additional eradications are planned. We review the eradications, both successful and unsuccessful, that have occurred in the United States. Most eradications involved the use of the anticoagulant rodenticides diphacinone and brodifacoum. Rodenticides have been applied by hand-broadcast, bait station deployment, and aerial broadcast. We briefly review the strategies and methods used in eradication projects and the efforts to mitigate potential non-target and environmental impacts.

Keywords: *Rattus*, rodent, rodenticide

INTRODUCTION

Introduced rodents pose a serious threat to the native flora and fauna of islands (Moors and Atkinson 1984; Veitch and Clout 2002; Engeman *et al*. 2006). Rodents can be prolific on islands where they have few or no predators. Their omnivorous foraging has led to the endangerment or extinction of numerous island species (Moors and Atkinson 1984; Witmer *et al*. 1998; Veitch and Clout 2002; Engeman *et al*. 2006,). Most seabirds that nest on islands have not evolved to deal with mammalian predators and are very vulnerable to introduced rodents and other species introductions. There has been a concerted worldwide effort to eradicate introduced rodents from islands with numerous successes (Howald *et al*. 2007). These efforts have relied heavily on the use of rodenticides (Howald *et al*. 2007; Witmer *et al*. 2007c). In this paper, we review the strategies and methods used and success with rodent eradications from islands in the USA. We also provide the first comprehensive list of attempted eradications.

INVASIVE RODENT INTRODUCTIONS AND DAMAGE

Many species of terrestrial vertebrates have been introduced into the United States and its territories (Witmer *et al*. 2007b; Witmer and Fuller 2011). The most common introductions are the commensal rodents, which have been widely introduced around the world (Long 2003). They include: Norway (*Rattus norvegicus*), ship (*R. rattus*), and Pacific (*R. exulans*) rats and two subspecies of house mouse (*Mus m. musculus, M. m. domesticus*). Other non-native rodents that have been introduced include nutria (*Myocastor coypus*, Carter and Leonard 2002) and Gambian giant pouched rats (*Cricetomys gambianus*; Engeman *et al*. 2006). Species native to the mainland and introduced to some islands include Arctic ground squirrels (*Spermophilus parryii*, Ebbert and Byrd 2002) and hoary marmots (*Marmota caligata*, United States Department of Interior 2010). It is possible that there have been undocumented introductions of other native rodents (deer mice, *Peromyscus* spp., and voles, *Microtus* spp.) to some islands for research purposes. Long (2003) reviewed the many rodent introductions around the world.

Rodents were introduced to islands for a variety of reasons and by various pathways. Most arrived accidentally as a result of shipping, shipwrecks and inadvertently landed with stores by landing parties. Some, possibly including hoary marmots, were introduced as a source of subsistence food for people. Other species, such as Arctic ground squirrels, were introduced as a food source of foxes that were introduced to islands for fur harvest. Nutria were introduced to numerous states and islands for the fur industry. Gambian giant pouched rats were introduced indirectly as escapes from the pet industry (Long 2003; Engeman *et al.* 2006).

Several types of damage have been caused by rodent introductions to the United States (Hyngstrom *et al*. 1994). A major impact is harm to native flora and fauna, including species endangerment and extinction with implications for ecosystem structure and function. In some cases, such as in the Hawaiian Islands, there has been substantial damage to agriculture, including crops in the field and stored foods. Rodents are also responsible for disease hazards such as plague and monkeypox (Meerburg *et al*. 2009).

PLANNING CHALLENGES

Planning and conducting a successful invasive rodent eradication from islands poses many challenges and should not be undertaken without a thorough commitment and adequate resources. The basic tenets of a successful eradication are: all individuals must be put at risk; animals must be removed faster than they can reproduce; and the risk of immigration must be zero (Parkes and Murphy 2003).

An eradication attempt that is 99% successful can ultimately result in 100% failure. Because of the large commitment of resources and public funds in eradication efforts, the potential for failure should be minimised. At times, as was the case with the giant Gambian pouched rat population in the Florida Keys, there was inadequate knowledge about the ecology of the invasive species in its newly invaded "habitat" (Witmer and Hall 2011). Obstacles to success can include inadequate funding and public support. Many people are sensitive to – or even strongly opposed to – the use of chemicals and lethal methods on public lands. People and non-target animals may disturb or damage traps or bait stations. Refused access to properties can be an impediment to eradication. People may provide food and water outdoors for pets or for feral cat colonies that then becomes available to the invasive

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rodents. Monitoring rodent populations when they are at low density is problematic. This presents the difficulty of detecting a newly-arrived invasive rodents or completing the final (and necessary) "mop-up" operation to get the last few rodents in an eradication effort. These issues make the achievement of a successful invasive rodent eradication a real challenge, especially in inhabited areas.

Agency reports and some personnel communications suggest that early eradication attempts in the USA involved relatively little planning or situation evaluation. In recent years, there has been more extensive planning, more pre-eradication monitoring of invasive rodent populations and potential non-target animals (especially threatened or endangered species), and increased efficacy testing of methods and rodenticides. Additionally, environmental assessments are now completed to assure that the proposed action is justified, in compliance with state and federal laws and regulations, and that the hazards to the environment and non-target animals will be minimal or adequately mitigated. Public involvement and support are usually incorporated as well. The steps involved in planning and implementing a robust eradication strategy with a high probability of success involves:

Preliminary monitoring and research
Feasibility of eradication
Regulatory compliance
Public information and communications media
Public support
Technical assistance and operations
Planning
Logistics
Procurement of equipment and other services
Monitoring and research
Staff recruitment and training
Implementation
Contingency planning
Follow up monitoring
Implementation of a bio-security plan

RODENT ERADICATIONS

We learned of 40 rodent eradication attempts in the United States and its territories (Table 1), some of which were on clusters of islands (e.g., Midway Atoll, Anacapa Islands, Bay of Islands). Most historic attempts were not well documented, so some may have been overlooked. The list is considerably longer than one presented by Howald *et al*. (2007), mostly because of an increase in the rate of attempts in recent years (e.g., 12 since 2004).

Of the 40 attempted eradications, 22 (55%) were successful (Table 1). For some failed attempts, it is difficult to know if the eradication failed or there was a relatively rapid reinvasion. This can be the case when target islands are near others that still have rat populations capable of natural dispersal. This was recently documented by Russell *et al*. (2005) in which case a radio-collared Norway rat swan 400 m from one island to another. This ability of rats may have affected eradication success in the Bay of Islands (Dunlevy and Scharf 2007). Molecular genetics have become a powerful indicator of whether the reappearance of rodents has been in response to a failed eradication or a subsequent re-invasion. For example, analyses of rat DNA on Congo Island suggests that rats found on the islands shortly after an eradication attempt were probably survivors, not invaders (Antoinette Piaggio pers. comm.). The 2-year rule of thumb is frequently applied after eradications: if no rodents are detected for the following 2 years with relatively intensive monitoring, the eradication can be considered successful (Howald *et al*. 2007; Witmer *et al*. 2007c).

Just over half (about 55%) of the islands were less than 20 ha. Some larger islands have been cleared of rats in recent years (e.g., Rat Island; 2900 ha). Aerial broadcast baiting has allowed the larger islands to be attempted more efficiently. Now that many of the methods and logistics of conducting island rodent eradications in the United States have been worked out and numerous successes achieved, we can probably expect more successful eradications. Planning for other island rodent eradications is already under way.

Approaches to Rodent Eradications

About 27 island eradications (67.5%) of rodents in the United States used the first generation anticoagulant diphacinone (0.005% active ingredient). In contrast, worldwide island rodent eradications most commonly used the second generation anticoagulant brodifacoum (Howald *et al*. 2007). Only nine eradications on islands (22.5 %) in the United States used brodifacoum (0.0025% active ingredient). In at least two cases, both diphacinone and brodifacoum were used and in a few cases bromethalin or bromadialone were used, but only in conjunction with brodifacoum. Currently, the USDA Animal and Plant Health Inspection Service (APHIS) has two rodenticides registered with the United State Environmental Protection Agency (EPA) for island conservation purposes: one formulation of diphacinone pellets and two formulations of brodifacoum pellets (Witmer *et al*. 2007c).

Most eradications (about 75%) used bait stations, often in conjunction with some hand broadcasting of baits. Hand broadcasting was usually in cliff areas and/or dense vegetation thickets. In recent years, there has been a trend towards aerial broadcast of rodenticide pellets from helicopters, using calibrated buckets and GPS guidance systems to help assure complete island coverage (Howald *et al*. 2005). The APHIS rodenticide registrations for conservation uses have allowed this to become more commonplace.

Reducing Non-Target Species Hazards

Rodenticide use poses risks of primary hazards through direct consumption and secondary hazards through the consumption of poisoned animals. Substantial efforts are made to minimise the loss of non-target animals which are often the resources that eradications of rodents aim to protect. On many islands, the risks to non-target mammals from rodenticide use are non-existent or very low because there are few, if any, species of native terrestrial mammals. The main safeguard for the safe use of rodenticides in conservation efforts is carefully following the EPA-approved label instructions for the product. Other basic considerations include the rodenticide product used; when, where, how and how much of it is applied; cleaning up spills promptly; and not using rodenticides in areas where there are highly valued or protected wildlife, as determined by pre-operation monitoring.

Other mitigation measures used in island eradication efforts are often selected on a case-by-case basis. The timing of bait application (especially with broadcast baiting) may be done after migratory birds have left the island to reduce their chance of direct or indirect exposure (Howald *et al*. 2005). Bait pellets can be large enough to help assure that they will not be consumed by small granivorous birds and pellets coloured dark green or blue can reduce their visibility to birds and lizards. Specially-designed bait stations can be used to restrict access by non-target species (e.g., Witmer *et al*. 2007a).

Table 1. Invasive rodent eradications in the United States with question marks denoting projects that need additional monitoring to confirm a successful eradication.
Species: e = *Rattus exulans*, r = *R. rattus*, n = *R. norvegicus*, m = *Mus musculus*, C. = *Cricetomys gambianus*, y = *Myocastor coypu.*
Toxins: brod = brodifacoum, brom = bromethalin, broa = bromadialone diph = diphacinone, zinc = zinc phosphide.
Methods: b = bait stations, h = hand broadcast, t = traps, a = aerial broadcast, sn = snares, sh = shooting.
Status: Y = successful, F = failed, R = reinvasion

Region Island	Area (ha)	Spp	Year Erad.	Toxin	Method	Status	Reference
Pacific Ocean							
Rose Atoll, American Samoa	6	e	1990-92	brod, brom	b, t	Y	Murphy and Ohashi 1993
Palmyra I., Line Islands	230	r	2001	brod	b	F	Howald et *al*. 2004
Cocos I., Guam	33.6	e, m	2009	brod, diph	b, t, h	Y?	Lujan pers. comm.
Midway Atoll Spit & Eastern, HI	134	r	1994-95	brod, brom	t, b	Y	Murphy, unpubl.
Kure Atoll, HI	105	e	1993	brod, brom	t, b	Y	Murphy, unpubl.
Mokoli'i I., HI	1.5	r	2002	diph	t, b	Y	Smith e*t al*. 2006
Mokapu I., HI	4	e	2008	diph	a	Y	Dunlevy pers. comm.
Lehua I., HI	125	e	2009	diph	a	F	Dunlevy pers. comm.
Anacapa Is. (3), CA	296	r	2001-02	brod	a, h	Y	Howald e*t al*. 2005
Bering Sea							
Rat I., AK	2900	n	2008	brod	a	Y	Howald pers. comm.
Bay of Islands, AK (12 I.)	0.1-17.8	n	2003	diph	b, h	most F or R?	Dunlevy and Scharf 2007
Caribbean Sea							
Monito I., PR	15	r	1993, 1998-99	brod, broa	b, h	1st F, 2nd Y	Garcia e*t al*. 2002
Steven Cay, USVI	0.8	r	1983	diph	h	Y	Pierce pers. comm.
Dog Cay, USVI	4.8	r	1983	diph	h	Y	Pierce pers. comm.
Kalkun Cay, USVI	1.4	r	1982	diph	h	Y	Pierce pers. comm.
Ruth Cay, USVI	14	r	2007	none	t	Y?	Pierce pers. comm.
Green Cay, St. Croix, USVI	5.2	r	2000	none	t	Y?	Pierce pers. comm.
Buck I., St Croix, USVI	72.7	r	1999-00	diph	b, h	Y	Witmer e*t al*. 2007a
Dutchcap Cay, USVI	12.9	r	2004	diph	b, h	Y	Pierce 2007
Saba I., USVI	12.3	r	2003	diph	b, h	Y	Pierce 2007
Capella I., USVI	9	r	2005	diph	b, h	Y	Pierce 2007
Buck I., St. Thomas, USVI	16.8	r, n	2005	diph	b, h	Y	Pierce 2007
Congo Cay, USVI	10.6	r	2004, 2006	diph, brod	b, h	both F	Hall *et al.* 2006, Pierce 2007
Gulf of Mexico							
Egmont Key, FL	112	r	2009	diph	b, h	Y	Hall pers. comm.
Grassy Key, FL	400	c	2007-cont	zinc	b, t	F	Hall pers. comm.
Chesapeake Bay							
Blackwater NWR	5200	y	2004	none	t, sn, sh	Y?	Kendrot and Sullivan 2009

Raptors and/or scavengers have sometimes been taken into captivity or temporarily relocated to reduce their exposure to animals consuming the bait (Howald *et al.* 2005). Endemic species of rodents can be held in captivity and a breeding colony can even be established. Collecting and removing or burying rodent carcasses can reduce risks of secondary poisoning, but often few carcasses are found because many rodents die underground. If single aerial broadcast-baiting with brodifacoum pellets is effective for rodent eradication then that approach may reduce the time bait is available to non-target animals versus repeated placement of bait by hand or in bait stations or several broadcasts. In the United States, generally two aerial bait drops are used to help assure a successful eradication. Valued or protected animals on some islands may require that bait is not placed in some areas (e.g., enclosures or pens); in these cases, invasive rodents are removed from the bait-protected areas by the use of live-traps or other means. Similar measures may also be instigated to protect fresh water bodies from bait ingress. Extra diligence must be exercised when threatened or endangered species are present as these species are protected under federal and/or state laws (e.g., Endangered Species Act, Migratory Bird Protection Act).

In general, impacts to non-target species during invasive rodent eradications should be considered in terms of population-level effects, rather than the effects to individuals, and in terms of the "greater good' that is achieved from a successful eradication. While there will probably always be some losses of non-target animals, proper precautions should minimise such risk and allow for the rapid recovery of affected populations (Howald *et al.* 2005). Those involved with successful invasive rodent eradications on islands are often surprised at how rapidly the island's flora and fauna recover after rodents are removed (Witmer *et al.* 2007a).

CONCLUSIONS

Seabird populations, sea turtle populations and other island resources warrant protection from invasive rodents. The recovery of fauna and flora on uninhabited islands after a successful rodent eradication is particularly notable (Witmer *et al.* 2007a). The significant impacts of introduced rodents on native flora and fauna have been repeatedly demonstrated. Invasive rodents are very adaptable, can exploit a wide array of resources as food and cover, and can increase reproduction very quickly when and where abundant resources exist (Macdonald *et al*. 1999). While invasive rodents will continue to pose challenges to land and resource managers, they can be controlled or even eradicated with a well-planned and adequately-supported effort using rodenticides. With proper planning, non-target losses will be minimal and these populations, along with other island resources, will often recover quickly after the rodents have been removed.

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New Techniques

New technologies planned, tested, and used for plant and animal eradications; and new approaches to eradications, such as dealing with multiple invasive species.

Successful control of an incipient invasive amphibian: *Eleutherodactylus coqui* on O‘ahu, Hawai‘i

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Abstract A Puerto Rican icon, the coquí frog (*Eleutherodactylus coqui*) has quickly proliferated across the Hawaiian Islands (Hawai‘i) since its introduction in the late 1980s. Shipping of goods, particularly commercial plants, provided coquí with easy passage between islands. Coquí are now firmly established on the Island of Hawai‘i (Big Island) and are the subject of early detection, eradication, and control activities on O‘ahu, Kaua‘i, Maui, and Moloka‘i. Hawai‘i provides an ideal home for coquí; all the benefits of its tropical native range and none of its natural predators. Large coquí populations threaten native arthropods and pose serious problems for the tourism and real estate industries. On O‘ahu, coquí distribution is sparse; only one naturalised population has been documented, in the town of Wahiawā, between a military base and a residential neighbourhood. A multiagency coordinated response resulted in successful eradication. The agencies involved include the O‘ahu Invasive Species Committee, the Hawai‘i State Department of Agriculture, the Department of Land and Natural Resources, and the U.S. Army Garrison Hawai‘i, collectively known as the Coquí Frog Working Group (CWG). Four elements were essential to success: 1) a control method permitted by federal regulatory agencies was known and available; 2) control crews were allowed complete access; 3) there was adequate funding for the operation; and 4) the population was relatively small. After close to a decade of work, the Wahiawā population was eradicated using a combination of habitat modification, nighttime citric acid vegetation sprays, and daytime citric acid ground drenches. Wahiawā is the first such eradication documented in the State.

Keywords: Invasive alien species, *Eleutherodactylus coqui*, coquí, amphibian, alien species control, Hawai‘i, O‘ahu

INTRODUCTION

The Puerto Rican frog, *Eleutherodactylus coqui*, or coquí, has become a major pest in the Hawaiian Islands (Hawai‘i) (Hara *et al*. 2008). Initially reported in the late 1980s and early 1990s from isolated locations on the islands of Hawai‘i (Big Island) and Maui, coquí rapidly spread across the Big Island; coquí reached the island of O‘ahu by 1998 and Kaua‘i by 2001 (Kraus *et al.* 1999; Kraus and Campbell 2002). The horticultural trade is believed to be the source of the original infestations, as well as the primary means of dispersal within the State (Kraus *et al.* 1999). Kraus and Campbell (2002) documented several instances of intentional dispersal by Hawai‘i residents, although this accounts for only a small part of its spread. Worldwide, introductions of coquí have been documented in various Caribbean Islands, Guam, Florida, and other locations in the mainland United States (Austin and Schwartz 1975; Schwartz and Henderson 1991; Joglar and Rios-Lopéz 1998; Christy *et al.* 2007; Rodder 2009). Climate envelope modeling by Rodder (2009) suggests that coquí could thrive across almost all the world’s tropical landmass.

In 1999, Kraus *et al*. considered that Hawaiian coquí populations were small enough to be eradicated, but four years later opined that this was now impossible (Kraus and Campbell 2002). Although coquí were detected just as populations entered logarithmic growth in 1997, agencies were unsuccessful in halting its spread (Kraus and Campbell 2002). Typically, new infestation sites in Hawai‘i require only a year or two to establish growing populations (Woolbright *et al.* 2006). Coquí have been reported from six of the eight main Hawaiian Islands (Fig. 1). Control efforts are ongoing across the State and are detailed in the draft Hawai‘i’s Coquí Frog Management, Research and Education Plan (Anonymous 2008). Single calling males heard on Moloka‘i and Lana‘i were removed and both islands are considered coquí-free. Seven locations on Kaua‘i had coquí; control efforts are ongoing at the only naturalised population (Anonymous 2008). Across Maui, coquí have been reported from 400+ sites. Most of these were single frogs that have been removed. One site (Honopou) similar in size to Wahiawā was deemed coquí-free in July 2008 (Anonymous 2008, Radford pers. comm.). The largest remaining naturalised site encompasses 91 ha of steep gulch (Radford pers. comm.). On the Big Island, managers feel that coquí eradication is no longer feasible. Despite efforts by concerned communities and government agencies, coquí are found in almost every lowland district. In the Puna district alone, coquí have spread over 17,000 ha (Anonymous 2008).

On O‘ahu, most reported coquí locations have been at nurseries or residences; all can be attributed to horticultural or other cargo goods from the Big Island and are the focus of localized control efforts. Wahiawā was the only known naturalised population in a wild setting. In 2006, we successfully eradicated the Wahiawā population (Fig. 1).

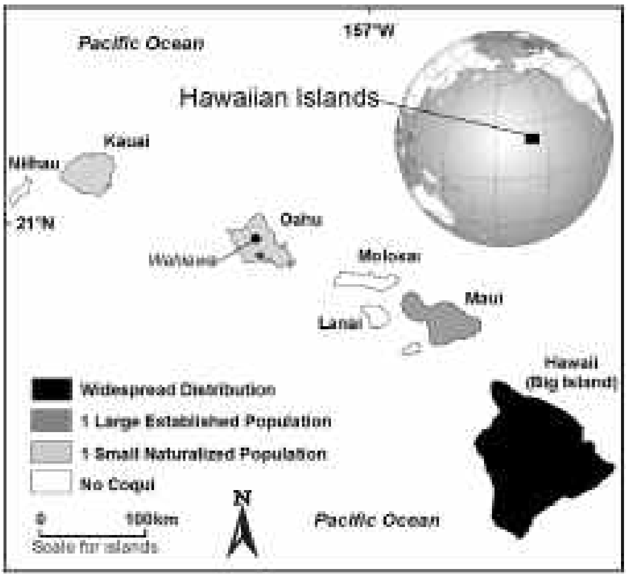

Fig. 1 *Eleutherodactylus coqui* distribution in Hawai‘i.

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We review the biology of coquí, their impacts in Hawai'i, and control methods. Our success can be attributed to four factors: the small geographical size of the infestation, the existence of tested control methods, the cooperation of landowners, and consistent funding.

Coquí are small (30-52mm), cryptically coloured, and easily overlooked (Schwartz and Henderson 1991; Campbell 2000). Eggs are fertilised internally and undergo direct development, with no tadpole stage and no need for standing water (Townsend *et al.* 1981; Townsend and Stewart 1985). Coquí eggs are susceptible to desiccation and are laid directly into protected nests, such as rolled leaves on or close to the ground, which are then tended by male frogs until several days after hatching, 17-26 days (Townsend *et al.* 1981, Townsend 1989; Schwartz and Henderson 1991). Coquí of all size classes thrive in wet environments, although they can adapt to dry periods via behavioural changes which reduce exposed skin surface area (Rogowitz *et al.* 1999).

In Puerto Rico, coquí densities average 20,570 individuals per hectare, among the highest recorded for any vertebrate, and provide a bountiful food supply for spiders, other invertebrates, birds, frogs, and snakes (Schwartz and Henderson 1991; Stewart and Woolbright 1996; Woolbright *et al.* 2006). Coquí hide in retreat or nest sites during the day, becoming active at night, when they call, mate, and feed in leaf litter, understory, and canopy, consuming up to 114,000 arthropods per hectare in a given night (Woolbright 1985; Stewart and Woolbright 1996). The number of protected nest and diurnal retreat sites limit population size (Stewart and Pough 1983). Males climb 1.5 – 3 m into the understory to make their characteristic loud "co-qui"call, which is between 90-100 decibels at a distance of 0.5 m (Narins and Hurley 1982; Rodder 2009). Mating activity peaks in warm summer months; this is when coquí presence is most obvious to observers (Woolbright 1985; Townsend and Stewart 1994). The unique biological characteristics and fecundity of coquí highlight the need for early detection and control.

In Hawai'i, coquí population densities may be several times higher than in Puerto Rico, with 28,000 to 89,000 frogs per hectare documented on the Big Island by Woolbright *et al.* (2006) and 91,000 frogs per hectare recorded by Beard (2008) at Manukā Natural Area Reserve. These high densities are likely due to an abundance of appropriate habitat, including retreat sites, and a lack of predators. Dense Hawaiian populations are theorised to consume proportionally higher quantities of prey than Puerto Rican populations (Beard 2007). This raises alarm in Hawai'i, where a unique biota evolved over millions of years, resulting in an extremely high rate of endemism and an ecosystem susceptible to invasion by unconstrained taxa (Loope and Mueller-Dombois 1989). Coquí target the most abundant prey available. In lowland, alien taxa-dominated ecosystems, they feed predominantly on non-native Hymenoptera and Amphipoda, but avoid termites and mosquitoes, common pests (Beard 2007). Native arthropod families most susceptible to coquí predation in the lowlands include Acarina, Coleoptera, Collembola, and Diptera (Beard 2007). Snails are also consumed by coquí, with approximately 12 endemic species documented in coquí stomach contents (Beard 2007). This is of considerable concern on O'ahu, which has at least 52 species of endemic terrestrial snails, including 24 listed as endangered by the federal government (USFWS 1999).

Currently, most coquí infestations are below 500 m elevation, while most remaining native taxa are above 500 m. When coquí reach native-dominated landscapes, they could have detrimental effects on under-surveyed endemic invertebrate and gastropod communities. Coquí may also compete for prey with native forest insectivores (birds, insects, bats) at upper elevations (Beard and Pitt 2005). Recent studies in Puerto Rico and Hawai'i indicate that coquí exhibit top-down ecosystem effects by increasing soil nutrient availability. Hawaiian ecosystems are naturally nutrient poor; generally, increased soil fertility favours invasive species over native species, thus facilitating additional invasion (Sin *et al.* 2008; Beard *et al.* 2002).

High densities of coquí could bolster numbers of mongoose (*Herpestes javanicus*), a predator of native birds, thereby indirectly increasing predation on native avifauna (Kraus *et al.* 1999; Beard and Pitt 2006). On the Big Island, Beard and Pitt (2006) found that coquí formed 19% of mongoose diet, although another important bird predator, the rat, consumed no coquí. These results are consistent with findings in Puerto Rico.

At natural densities, the musical chorus of the coquí is a beloved part of the Puerto Rican night. In Hawai'i, high densities of frogs create loud choruses which can exceed noise pollution standards set by the State Department of Health, affect residents' sleep, depress real estate sales, and impact tourism (Kaiser and Burnett 2006). The horticulture industry is most seriously affected. Cleaning contaminated nurseries and plants for safe shipping both intra- and inter-State adds to the basic cost of business (Raloff 2003; Kaiser and Burnett 2006). Perceptions of the frogs vary, with a small group of residents welcoming coquí, while others volunteer with control efforts (Kraus and Campbell 2002; Anonymous 2008).

STUDY AREA AND METHODS

Wahiawā coquí infestation site description

The infestation at Wahiawā (Fig. 2) was reported by a resident in 2001, after a small backyard nursery business unknowingly imported contaminated plants from the Big Island. This property borders Schofield Barracks East Range (SBE), which is managed for Army training by the United States Army Garrison Hawai'i (USAG-HI). Coquí dispersed into SBE and neighbouring residences, eventually colonising approximately 5.6 ha. The infestation site is a patchwork of residences/yards, small gulches, and highly structured, dense, alien forest. There are almost no native Hawaiian taxa in the region. A multi-agency partnership was established to develop a control strategy for Wahiawā and O'ahu in general. The Coquí Working Group (CWG) is made up of the O'ahu Invasive Species Committee (OISC), the Hawai'i State Department of Agriculture (HDOA), the

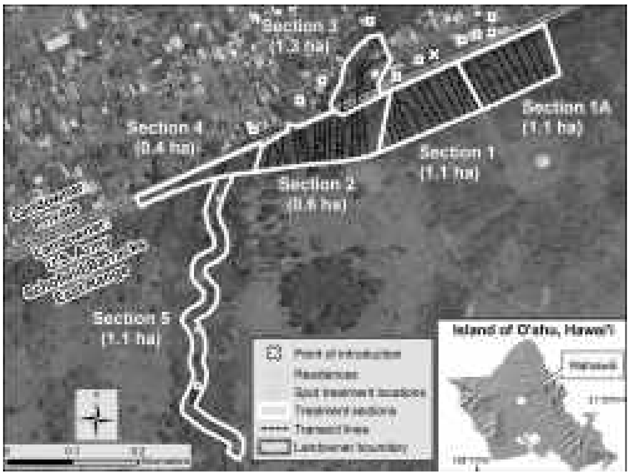

Fig. 2 Wahiawā coquí infestation area.

Department of Land and Natural Resources (DLNR), and the USAG-HI, which includes the Directorate of Public Works (DPW) and the O'ahu Army Natural Resources Program (OANRP). The Fish and Wildlife Service (FWS) also assisted with initial response efforts.

To facilitate control, the Wahiawā infestation was divided into six geographic treatment areas, or Sections (Fig. 2). The size of the infestation changed over time, requiring the creation of new Sections. Section boundaries were chosen to take advantage of natural landmarks, barriers, changes in terrain, and landowners. Small areas of 0.4-1.3 ha were found to be logistically easy to manage. Vegetation included canopy (>7 m) trees such as *Eucalyptus robusta*, mid-level trees *Citharexylum caudatum*, *Clusea rosea*, *Psidium cattleianum*, *P. guajava*, and *Schinus terebinthifolius*, shrubs such as *Clidemia hirta*, various heliconia and gingers, grasses *Melinis minutiflora* and *Urochloa maximum*, and several climbing Araceae. Excluding Section 3, private residences with reported coquí were not included in Sections. Control was conducted at these residences; data was tracked by address and is grouped here under the heading Residences. The CWG implemented an adaptive management plan emphasising habitat modification and chemical sprays to control coquí.

Habitat modification

In general, habitat modification was implemented first at each Section. Since coquí density is related to available nest and retreat sites, removing these sites reduces the carrying capacity of an area (Stewart and Pough 1983). Though highly labour intensive, it also results in less surface area to spray, facilitates the use of spray equipment (such as long hoses), and speeds spray operations. In Sections 1, 1A, and 4, habitat modification involved manual clearing and chipping of all/most understory vegetation. Cut vegetation was treated with triclopyr and glyphosate to prevent re-growth. Chips were left on site and monitored for calling frogs. Canopy trees and some understory were left in place to discourage frog emigration. In Section 3, the landowner bulldozed the area in preparation for building. This timely coincidence eliminated all vegetation and coquí habitat. No clearing was conducted on private residences or in Section 5. In Sections 1, 1A, 2, and 3, transects approximately 10 m apart were cleared and marked to facilitate sprays, monitoring, and data tracking (Fig. 2). Before seasonal spray operations began each year, the transects were maintained and sprayed with herbicide. Transects in Sections 1 and 1A were drivable (cleared with a bulldozer), while those in Section 2 were accessible on foot.

Chemical sprays

Chemical sprays followed vegetation removal. The United States Department of Agriculture's (USDA) Animal and Plant Health Inspection Service, Wildlife Service, and National Wildlife Research Center tested a variety of chemical coquí control techniques (Campbell *et al.* 2001; Pitt and Sin 2004; Pitt and Doratt 2006). Citric acid, lime, and caffeine were most effective against coquí (Pitt and Sin 2004; Pitt and Doratt 2006). Due to permitting issues, we opted not to use caffeine or lime (Pitt and Doratt 2005). Only citric acid was sprayed at Wahiawā. During initial spray operations, monitoring did not detect any significant non-target effect to the arthropod community. Citric acid at 16% solution is effective on all size classes of coquí and reduces hatching rates, although humidity levels can reduce efficacy in some cases (Doratt 2008). Citric acid is not regulated by the Environmental Protection Agency and is considered a minimum risk pesticide (Anonymous 2008). Frogs must be sprayed directly; citric acid residue alone is not an effective control (Hara *et al.* 2008). Two week intervals between citric sprays are recommended by Beard to allow for hatching of any surviving eggs (Beard and Pitt 2005). Beard also recommends spraying a given area at least three times to achieve eradication. We aimed to spray or monitor each Section two to three times per calling season, roughly March through September. Sprays fell into one of two categories, area drench or hot-spot. Sprays were not conducted on rainy days.

Area drench spray, citric acid 16%

Typically, the first spray of a Section was an area drench spray. From 2003 to 2005, night area sprays targeting active frogs were conducted. A truck-mounted, motorised sprayer with a 380 L tank, 2 cm diameter hose, and tee-jet spray gun was used. Staff walked or drove along the cleared transects and sprayed at a rate of 53 L/min into the canopy, as high as possible (5-7m), coating every surface in the understory. Ground substrates were not intentionally saturated, except via the large amount of runoff from vegetation above. Care was taken to spray from multiple angles so as to achieve deep penetration of vegetation. After 2006, we switched to day drenches, targeting inactive frogs hiding in the leaf litter. We upgraded to a motorised sprayer with a 1515 L tank, 4 cm diameter fire hose, and brass adjustable fire nozzle with a 95 L/min spray capacity. Staff walked or drove along the cleared transects and drenched the ground and understory to 1 m in height. Large volumes of citric acid were applied; while hiding frogs are more difficult to spray than active, moving frogs, the CWG theorised that the high volume of spray would penetrate retreat sites. Logistically, daytime drenches allowed staff to work longer hours, treat more area, and reduce impact on residents.

Hot spot spray, citric acid 16%

Hot spot sprays focused on isolated calling males and were conducted during mop-up operations, following area sprays, and at private residences where area sprays were inappropriate. Generally, at least two weeks passed between initial treatment and mop-up. Working at night, staff identified small areas, often less than 5 × 5 m, with one or more calling frogs and sprayed the areas thoroughly. All materials from the height of the calling frog to the ground were drenched to the point of run-off. Both backpack hand-pump sprayers (11 or 19 L capacity) and motorised 380 or 1515 L tank sprayers were used. Hot spot sprays also treated females responding to calling males and any sub-adults in the spray zone.

Hand capture

Hand capture is not effective in naturalised coquí infestations, as non-calling females and juveniles are much less likely to be caught than calling males. Hand captures were conducted during initial work at Wahiawā in 2001-2003 before other techniques became available, opportunistically during monitoring trips, and at private residences. Some private citizens with few calling frogs preferred hand capture, as chemical sprays may result in burned vegetation. The technique is simple but requires time, skill, practice, and is most effective at night. Staff identify the general location of a calling frog, circle the calling perch to pinpoint its location, approach it from the rear, and place a clear plastic tube around it. The frog usually reacts by jumping into the tube. Captured frogs were generally preserved in alcohol or frozen.

Audio monitoring

Mark-recapture is the only method shown to produce precise estimations of coquí population density (Funk *et al.* 2003). Given the relatively small population size at Wahiawā (125 calling males) and the goal of eradication, we opted for less rigorous audio survey monitoring. This technique allowed us to establish presence/absence of frogs year-round, note any trends in numbers of calling males, identify locations of calling males and direct spray efforts towards these locations. Ideal audio survey conditions are temperatures above 17˚C, relative humidity above 60%, and recent rain. Evenings in Wahiawā often meet these requirements. We surveyed during peak calling time, between 5:30pm and 11:00pm. Staff walked through the infestation, listening and recording the number of calls/ estimated number of callers heard in each Section. Calling sites were mapped. Weather conditions were noted. Time spent monitoring was not consistent. The same set of observers typically conducted all audio surveys. Over the inactive winter season, monitoring was conducted once a month. Over the active summer season, monitoring was conducted twice a month, with incidental observations noted during control operations.

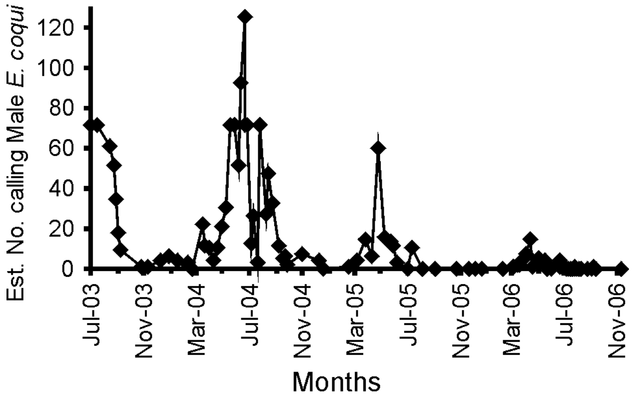

Fig. 3 Changes in numbers of calling male coquí at Wahiawā. Data prior to 2003 are excluded due to recording inconsistencies. From 2007 to 2009, all visits recorded no calling male coquí.

RESULTS

Successful eradication of coquí from Wahiawā required consistent, repeated control (Tables 1 and 2). Eradication took eight years, 2001 to 2008, although systematic control efforts did not begin until 2003. Spraying was conducted over five years, with two years of learning followed by three years (2005-2007) of full-scale efforts. Each Section was sprayed multiple times in a given year. The CWG conservatively assumed that sprays, which require direct contact of coquí with citric acid, would not necessarily penetrate to all nest/retreat sites, necessitating multiple treatments. Sprays were conducted for a full season after the last calling frog was heard. The final project year, 2008, was dedicated solely to monitoring. Over the course of control efforts, the infestation area increased five-fold.

At least two years of full-scale spray efforts, in conjunction with habitat modification, were required to achieve eradication in a given Section (Table 2). In the longest-infested Sections (1, 1A, 2, 3), spray efforts took four to five years, while in the shortest-infested Sections (4, 5), they took just two years. This difference reflects the steep learning curve of the CWG and the benefit of rapid response. At Sections 4 and 5, sprays began months, rather than years, after detection, allowing coquí less time to establish.

The 2006 upgrade to a larger capacity sprayer greatly reduced the effort required to treat a given area and increased the area treated per unit time. For example, in 2005 staff sprayed 55.6 L/person hour, while in 2006 staff sprayed 95.8 L/person hour. Note that person hours include mixing and some transport time. High volume equipment was vital in improving efficacy and facilitating thorough coverage across the entire infestation.

Terrain strongly influenced the time and volume of citric acid required for treatment (Table 2). At 0.6 ha, Section 2 was one of the smallest Sections. However, it required over three times the citric acid solution and person hours needed in either Section 1 or 1A, which are both over 0.4 ha larger. Sections 1 and 1A were completely flat, easy to clear, and easy to spray. Section 2 encompassed a small, steep-sided gulch. Less habitat modification was conducted at Section 2 as it was not possible to bulldoze transects, and it was more difficult to spray into the canopy. Fortunately, Section 5, which at 1.1 ha was twice the size of Section 2, and in a gulch with limited road access, only had coquí in four discrete locations and did not require treatment across its entirety.

Population size estimates from 2001 through 2003 are rough approximations, as data collection efforts were not yet standardised. Although it appears that coquí numbers increased from 2003 to 2004, despite control efforts, this likely is an artifact of inconsistent monitoring and not a true increase. However, between 2003 and 2005, numbers of calling males decreased in Sections 1 and 1A, but increased in Section 2, despite control. This again reflects the importance of terrain and high volume equipment in achieving efficient control.

After 2004, the number of calling males declined dramatically, which coincided with a concerted effort to conduct control on a regular, consistent schedule (Table 1). After 2003, the CWG realised that a dedicated crew was needed to accomplish the amount of control work required. The seasonal crew hired in 2004, despite working through logistical problems, demonstrated the efficacy of this approach. With the addition of a permanent supervisor, seasonal staff were able to treat the infestation multiple times in a season (Table 2).

The number of calling males consistently declined throughout the treatment period (Fig. 3). Each year, calling activity built through the summer, peaking in July and declining until October. Little or no calling was heard in the colder winter months, when coquí are less active. Summer peaks declined dramatically after 2004, when 125 males were heard on one evening in July. The last calling male was heard in September 2006 in Section 5. The Statewide plan on coquí management recommends that an infestation site be considered eradicated if no frogs are observed for at least one year. Coquí take ten months to develop from egg to mature adult (Stewart and Woolbright 1996). Given that frogs born in September 2006 would mature by June 2007, major spray operations continued through 2007. No frogs were heard in 2007 or 2008. After two years without detecting any coquí, the CWG felt confident that the population had been eradicated. One survey conducted in May of 2009 confirmed this, with no frogs heard.

There is a remote threat of reinvasion at Wahiawā. The plant nursery business which originally introduced coquí to the area continues to operate. Residents continue to purchase plants from vendors who import from the Big Island. Extensive outreach by the CWG educated residents and the nursery operator about coquí and garnered support for control operations. The CWG maintains its positive relationship with the community and regularly communicates with the nursery operator and other concerned residents. Preventing the establishment of new, naturalised coquí populations anywhere on O'ahu is a CWG priority; current efforts are directed at plant nurseries, early detection/rapid response, and public awareness.

DISCUSSION

Wahiawā is the first documented eradication of a naturalised coquí population in a wild site in Hawai‘i. This eradication was key to avoiding the establishment of coquí on O‘ahu, thus preventing a repeat of events on the Big Island. O‘ahu itself cannot be declared coquí free, as OISC and DOA continue to field reports of calling males from nurseries and residents who purchase plants from the Big Island. However, these response efforts require fewer resources than would be needed if coquí become established on O‘ahu.

Crucial to success at Wahiawā was the relatively small population size of the infestation, a federally approved control method, complete access to the infestation site, and adequate and consistent funding throughout control operations. If any one of these factors had not been in place, eradication of this small infestation would have been exceedingly difficult, if not impossible.

Population

The Wahiawā infestation was detected in the early phase of establishment in 2001, with initial estimates of approximately 100 calling frogs spread over 1.2-2.4 ha. When systematic control began in 2003, the area of the infestation had already increased to approximately 3.2 ha. By 2003, the infestation was poised to follow patterns observed on other islands: rapidly increasing density in infested zones, expanding infestation boundaries, and increasingly high noise levels. Despite active control efforts, in 2004 coquí spread east, necessitating the establishment of Section 1A (Fig. 2). Habitat modification efforts in 2003 may have encouraged frogs to seek new retreat sites in Section 1A. In 2006, coquí spread west,

Table 1 Chronology of control efforts at Wahiawā.

Date	Estimated Number of Calling Males	Control Activity Highlights	Control Techniques Applied
2001 May	≈ 100 +	Initial coquí report from a private residence. Population likely present for a year.	Hand capture
2001 May through 2003 June	≈ 100 +	Coquí spreads to Sections 1, 2, 3, Residences.	Hand capture
2003 June through 2003 Sept	≈ 50 +	Habitat modification in Section 1. Transects installed in Sections 1, 2. Public meetings held in Wahiawā with concerned residents. Nighttime area sprays begin in Sections 1 and 2. Hot spot sprays begin in Section 3 and Residences.	Habitat modification Area spray Hot spot spray Hand capture
2004 June through 2004 Nov	125+	Seasonal spray crew hired. Coquí spread into Section 1A. Habitat modification and transect installation in Section 1A. Nighttime area sprays continue in Section 2. Hot spot sprays conducted in Sections 1, 1A, 2, Residences Multiple problems encountered: delay in obtaining citric acid, high training activity in SBE by Army reduced access, sprayer procurement delayed.	Habitat modification Area spray Hot spot spray Hand capture
2005 Feb through 2005 Nov	60+	Seasonal spray crew hired. Transects in Area 1, 1A bulldozed to become drivable. Transects in Area 2 maintained. Nighttime area sprays conducted across Areas 1, 1A, 2, 3. Hot spot sprays conducted at Residences.	Habitat modification Area spray Hot spot spray Hand capture
2006 Mar through 2006 Oct	29	Permanent vertebrate supervisor, seasonal spray crew hired. Frogs spread to Sections 4 and 5. Habitat modification at Section 4. Section 3 bulldozed by landowner prior to development. Area sprays conducted across all Sections. Switch from nighttime sprays to daytime drenches. Hot spot sprays conducted across infestation. Only 29 calling frogs noted during course of season. Last calling frog heard Sept 06 in Section 5.	Habitat modification Area spray Hot spot spray Hand capture
2007 Feb through 2007 Nov	0	Seasonal spray crew hired. Daytime area sprays conducted at Sections 2 and 4. Hot spot sprays conducted at Sections 1, 1A, 5 and Residences. 1 year without calling frogs indicates population likely eradicated	Habitat modification Area spray Hot spot spray Hand capture
2008 Mar through 2008 Sept	0	No seasonal crew hired or control conducted. Monitoring primary focus.	None
2009 May	0	Over two years with no calling frogs. All efforts end. Wahiawā infestation eradicated.	None

into Sections 4 and 5. Both sections lie down a gulch of the original infestation. Exceptionally heavy rains in 2006 flooded the normally dry gulch and may have dispersed frogs downstream. While coquí have been shown to exhibit some homing behavior in Puerto Rico, if displaced more than 100 m they are unlikely to return to original nest sites (Gonser and Woolbright 1995). Homing behavior has not been studied in Hawai‘i, and coquí may behave differently in a predator-free, retreat-rich environment. On SBE, several wide dirt roads may have discouraged coquí from dispersing to the south. The infestation did not cross roads; rather frogs appear to have spread from yard to yard, into semi-wild areas, and along gulches. Although the area of the infestation increased in 2006, numbers of calling males drastically declined. Only 29 males were heard the entire season, with 15 of them heard in one night in May 2006. Due to the small, 5.6 ha size of the entire infestation, the CWG was able to track population expansion and direct sprays to where they were most needed. This flexibility was vital.

The draft Hawai‘i’s Coquí Frog Management, Research and Education Plan discusses a rough formula, based on Puerto Rican data, for translating numbers of calling males to population estimates (Stewart and Woolbright 1996; Anonymous 2008). Assuming that calling males and reproductive females are found at a ratio of 1:1, and pre-adults and adults are found at a ratio of 5.3:1, where X is the number of calling males, the population size equals $X \times 2 \times 5.3$. While prolific in Puerto Rico, under laboratory conditions coquí are even more fecund, suggesting that reproductive potential is elevated in Hawai‘i’s predator-free, wet environment (Townsend and Stewart 1994; Hara *et al.* 2008). Although this formula has not been tested against field data in Hawai‘i, it may underestimate Hawaiian coquí fecundity, and was not used to estimate population size during control operations, it

Table 2 Summary of spray effort by infestation section.

Section # (date of detection) Area (ha)		Year	Number of Area Drench Sprays	Number of Hot Spot Sprays	Citric Acid Solution (litres)	Person Hours	Date Last Coquí Heard
Section 1		2003[1]	1	-	757	9	
(2001)		2004[1]	-	3	1325	32	
1.09 ha		2005	1	-	15,369	276	July 2005
		2006	1	1	17,754	185	
		2007	-	1	1893	26	
	Total:	5 yrs	3	5	37,098	528	
Section 1A		2004[1]	-	3	1325	32	
(2004)		2005	2	-	14,385	288	
1.13 ha		2006	1	1	19,873	182	August 2005
		2007	-	1	378	12	
	Total:	4 yrs	3	5	35,961	514	
Section 2		2003[1]	1	-	3028	90	
(2001)		2004[1]	1	1	8290	96 + [2]	
0.61 ha		2005	2	-	29,148	784	August 2006
		2006	2	1	39,747	522	
		2007	1	-	39,368	480	
	Total:	5 yrs	7	2	119,581	1972+	
Section 3		2003[1]	-	1	757	INC[2]	
(2001)		2004[1]	INC[2]	INC[2]	INC[2]	INC[2]	
1.25 ha		2005	2	-	10,978	201.5	July 2005
		2006	1 partial	-	13,627	130	
	Total:	4 yrs	2 + partial	1	25,362	331.5	
Section 4		2006	3	-	16,092	134	
(2006)		2007	2	-	8706	95	May 06
0.45 ha	Total:	2 yrs	5	0	24,798	229	
Section 5 [3]		2006	1	3	14,006	134	Sept. 2006 [4]
(2006)		2007	2	-	8328	103	
1.05 ha	Total:	2 yrs	3	3	22,334	237	
Residences		2003[1]	-	1	378	2.5	
(2001)		2004[1]	-	1	1136	35	
0.9 ha		2005	-	4	2082	51	May 2006
		2006	-	8	14,536	244	
		2007	-	2	1514	44	
	Total:	5 yrs	0	16	19,646	376.5	
	Grand Total	5 yrs	N/A	N/A	284,779	4188	Sept. 2006

[1] Records from 2003 and 2004 combined Sections into single sprays. For these years, citric acid volumes and person hours are estimated based on areas and field notes.
[2] Data incomplete (INC)
[3] Four discrete sites with frogs in this Section. These were the only sites sprayed.
[4] Coquí heard 28 Sept. 2006. Spray conducted October 2006. No frogs heard during surveys in November.

provides an interesting picture of likely population size at Wahiawā. Using this formula and the best estimates in each year, the population was approximately 1060 frogs in 2001, increasing to 1325 in 2004, decreasing to 159 in May 2006, and sinking to 10 in September 2006. This exercise demonstrates that even small numbers of calling males may indicate sizable breeding populations. Audio monitoring is crucial to achieving eradication and the presence of any calling males requires immediate action.

Control Methods

Coquí were first recognised as a problem in Hawai'i in 1999 (Kraus *et al.* 1999), but no effective, legal control techniques were available until 2002 (HDOA 2001). Caffeine application was allowed in 2002 under a year-long federal special use permit, which was not renewed (Hara *et al*. 2008). Citric acid became available in 2003, and does not require a federal permit (Hara *et al*. 2008; Pickhardt and Redding 2002). Lime was not federally approved until 2005; the special use permit allowing its use expired in 2008 and has not been renewed (EPA 2005; Hara *et al*. 2008). Developing control methods and obtaining federal approval for them is time-consuming, but vital for success. The lack of an effective, approved control method hindered early control efforts across the State, with lasting repercussions for the Big Island (Anonymous 2008). At Wahiawā, the relatively short two-year lag between coquí discovery and development of citric acid sprays likely allowed the infestation to become established.

Access

Most of the Wahiawā coquí were located on Army training land, but low numbers of frogs were also present at numerous private residences. While Army training activities occasionally hampered operations, the USAG-HI cooperated with coquí control efforts, facilitating access to SBE for day and night operations. Although a few private residents were unsupportive, the neighbourhood as a whole was committed to eradicating coquí, providing regular access, allowing citric acid to be sprayed on their yards, and patiently dealing with noise from night sprays. OISC talked with local politicians and community boards to ensure support. In contrast, on the Big Island, a small group of coquí enthusiasts support the presence of coquí in the islands and are vocal in their disapproval of government-sponsored control (Beamish 2004). While the coquí is on the State Noxious Pest list, allowing for control without landowner permission, cooperative landowners and a supportive public smooth the way for effective control. On Kaua'i, complete eradication is hampered by one resident who does not allow any control work on her property (Gundersen pers. comm.)

Funding

The CWG was fortunate in having adequate funding throughout the entire Wahiawā operation, which conservatively cost US$279,113, excluding in-kind contributions of labour from CWG partners and OISC administrative time. Most of this cost went to labour and citric acid. Funding sources include the State of Hawai'i, the City and County of Honolulu, and the USAG-HI. In-kind contributions came from HDOA, DOFAW/NARS, and the USAG-HI. OISC took on primary responsibility in soliciting funding. On Maui, lack of adequate funding to mount the operation required at the 91 ha gulch population has been a major roadblock in achieving eradication (Penniman pers. comm.).

CONCLUSIONS

Given adequate resources and staffing, small coquí populations can be eradicated. It took the CWG eight years of effort to eradicate the Wahiawā infestation. In future, similar infestations should require much less time and resources. Essential elements for success included dedicated spray crew staff, an aggressive spray drench schedule, high volume spray equipment, major habitat modification, sufficient citric acid, and simultaneous control and monitoring across the entire infestation. The establishment of new coquí colonies on O'ahu can only be prevented by stringent inspection of Big Island imports and regular surveys on O'ahu. While coquí may very well be a permanent part of the Big Island landscape, the other main Hawaiian Islands need not follow the same path.

ACKNOWLEDGEMENTS

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Invasions and stable isotope analysis – informing ecology and management

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Abstract Stable isotope analysis has increasingly been used to answer ecological questions. However, despite their potential, stable isotopes have rarely been used to assist with managing invasions. Here we discuss some of the principles behind the use of stable isotope analyses. We also review how stable isotopes can aid our understanding of the threats posed by invasive species, and the mechanisms by which some species successfully invade new environments. We then show how information from stable isotopes can be used to evaluate and refine ongoing management actions from an early stage in eradication attempts. We highlight the potential for such approaches to rapidly and simply provide detailed ecological information. We conclude that this technique can be used not only to inform our understanding of the problems caused by invasive species, but also to facilitate conservation and resource management objectives for wildlife populations.

Keywords: adaptive management, invasive species, island, spatial marker, trophic web

INTRODUCTION

Stable isotope analysis (SIA) has increasingly been used to answer ecological questions about organisms, including those relating to diet and migration patterns (Dalerum and Angerbjorn 2005; West *et al.* 2006; Crawford *et al.* 2008; Inger and Bearhop 2008). This focus has in turn led to an increase in the number of chemical elements used and the variety of ecological questions addressed (Fry 2006). Finer resolution to SIA has been aided by recent advances in statistical modelling (Phillips and Eldridge 2006; Parnell *et al.* 2010; Jackson *et al*. 2011). Stable isotope analyses have only recently been applied to invasion biology. This short review examines how SIA can be an additional tool to assist with the management of invasive species. We first discuss the ways SIA can be applied, then show how it can assist with studies of invasion biology as well as refining approaches to eradication campaigns.

BASICS OF STABLE ISOTOPE ANALYSIS

Many chemical elements can have more than one isotopic form of differing molecular mass. Examination of the stable isotopic ratios of the various forms of oxygen, hydrogen, sulphur and strontium have all helped to provide unique insights into the ecology and biology of plants and animals (e.g., Crawford *et al.* 2008), but the two most commonly used are stable isotopes of carbon (C) and nitrogen (N). The former (the ratio of ^{13}C to ^{12}C, expressed as $\delta^{13}C$). can be used to discriminate among different sources of primary production. The derivation of such ratios can potentially identify an animal's foraging location. For example, there is a difference between marine and terrestrial sources, where marine signatures are enriched with ^{13}C compared to terrestrial ones (Fry 2006). Stable nitrogen isotope ratios, (^{15}N to ^{14}N, expressed as $\delta^{15}N$) can also vary spatially, but are much more useful as a means for determining the trophic level at which an animal is feeding. Organisms become progressively enriched in ^{15}N at higher trophic levels due to preferential retention of the heavier isotope during tissue synthesis (Fry 2006). As a result there is a stepwise style enrichment between consumer and prey, meaning that animals feeding at higher trophic levels within a food web have higher $\delta^{15}N$ in their tissues than those feeding at lower trophic levels..

With the increasing use of SIA, methods have been developed to quantify the importance of multiple food sources or determine the probability of animals moving across different localities. Such methods require stable isotopic ratios from the consumer's tissue (hair, feathers, whiskers, claw, blood, liver, bone etc), and also the stable isotope ratios of potential prey, local geology or rainfall patterns (e.g., West *et al.* 2006). Mixing models (Phillips and Gregg 2003) are often applied to interpret these results and, more recently, Bayesian solutions to these analyses have been developed (Parnell *et al.* 2010).

ADVANTAGES OF USING SIA

Many species are difficult or expensive to study in the field because of their behaviour or location. For example, the nocturnal and neophobic behaviour of rats (*Rattus* spp.), coupled with difficulties with accessing invaded islands can prevent detailed year-round study. Furthermore, the use of conventional techniques such as radio telemetry or scat analysis are a time consuming, labour intensive and costly method of measuring the ecological impacts of invasive species. By comparison, SIA is relatively cheap because time and effort required in the field can be reduced. This is because behavioural information can be gathered over multiple time scales through the analysis of multiple tissue types from an individual after a single capture event. Since different tissue types are replaced at different rates, the proteins within them will be synthesised at different times. The proteins then reflect aspects of the animal's ecology at these different points. For example, stable isotope signatures from liver cells reflect the animal's diet over previous days, those of muscle reflect the diet over preceding weeks to months, and those of hair for longer still (Kurle 2009). The length of time represented by isotopes in tissues also varies with the metabolic activity of the animal concerned. Replacement processes are more rapid in species with higher metabolic rates, so that, for example, mice have faster replacement rates than rats (MacAvoy *et al.* 2006).

Single tissues such as claws, whiskers and hair can also be used to derive a time series of past behaviour. The protein in these tissues is metabolically inert after it has been synthesised, so provides a continual record which can be 'read' along its length, going back in time the nearer a sample is to the distal end of the tissue (Bearhop *et al.* 2003; Cherel *et al.* 2009). In sum, SIA of multiple tissues of an individual animal can rapidly provide a detailed record of diet and potential foraging locations over different time scales.

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DEMONSTRATING THE IMPACTS OF INVASIVE SPECIES

The accumulated knowledge of a species' impacts elsewhere remains the best predictor of their likely effects at new locations (Simberloff 2003). However, a business case for the eradication or control of invasive species will often still require site-specific information.

Direct predatory impacts

Stable isotope analyses can be used to examine the diet of invasive predators. While this provides an integrated picture of an animal's diet, SIA cannot be used to differentiate between predated and scavenged food items. For example, Stapp (2002) demonstrated that ship rats (*Rattus rattus*) in the Shiant Islands, UK consumed seabird flesh, but could not demonstrate predatory behaviour from this result. Although proof of predatory behaviour may not necessarily be derived from SIA, it has advantages over conventional methods such as stomach content or scat analyses, which may over-represent indigestible material or under-represent items that leave little visual trace. The most informative approach can be to combine SIA with other methods to strengthen the conclusions that can be drawn (Hobson *et al.* 1999). For example, Harper (2007) used experimental removal of ship rats and a predatory bird, weka (*Gallirallus australis*), alongside isotopic and conventional diet analysis to examine the importance of sooty shearwater (*Puffinus griseus*) in each species' diet. Caut *et al.* (2008) used a combination of SIA, stomach contents and direct observations to reveal the impact of ship rats on breeding seabirds on Surprise Island, New Caledonia. They also showed how, in the absence of seabirds, rats switched prey to green turtle (*Chelonia mydas*) hatchlings. Hobson *et al.* (1999) were similarly able to demonstrate the seasonal importance of breeding seabirds in the diet of brown rats *R. norvegicus* on Langara Island, British Columbia, and also the extent to which different individuals relied on this resource. Such plasticity in the consumption of seabirds by individual brown rats was also found when they fed on least auklets (*Aethia pusilla*) at Kiska Island, Alaska (Major *et al.* 2007).

Impacts on trophic structure

The way that invasive species disrupt food webs and transform community structure has also been revealed through the use of SIA (e.g., Vander Zanden *et al.* 1999; Croll *et al.* 2005). Changes in trophic level can be seen through examination of changes in $\delta^{15}N$ within a species over time, as was demonstrated for the invasive carnivorous Argentine ant (*Linepithema humile*) in California. Initially, invading ants occupied a similar trophic level to those in their native habitats, where they fed on other ants. However, once established, the ants shifted to a lower trophic position as they consumed more plant material following severe reductions in native ant prey populations (Tillberg *et al.* 2007). At a whole-island scale, Croll *et al.* (2005) used SIA to measure the importance of marine nitrogen input from seabirds to Aleutian Island plant communities. They found that on islands where invasive arctic foxes (*Alopex lagopus*) had destroyed the seabird populations, plant communities were transformed from grasslands to shrub/forb communities because of reduced soil fertility.

Invasive plants can also modify food webs. Stable isotopes ($\delta^{15}N$) demonstrated that invertebrates that persisted within areas invaded by the *Spartina alterniflora x foliosa* hybrid in San Francisco Bay, USA were consuming this invader (Levin *et al.* 2006). However, other invertebrates such as amphipods, which are an important prey item for many predators, were less tolerant to habitat invasion as they did not consume the hybrid plant. Thus while the invasive plant structurally altered the ecosystem, its resources were not efficiently broken down, resulting in bottom up alteration of the food web (Brusati and Grosholz 2009).

Energetics modelling combined with SIA was used to demonstrate dramatic changes in a food web on the Channel Islands, California following the introduction of feral pigs (*Sus scrofa*) (Roemer *et al.* 2002). The pigs provided an abundant food resource for golden eagles (*Aquila chrysaetos*), which increased in number. Increased predation by eagles reduced the population of island fox (*Urocyon littoralis*). This in turn allowed island skunk (*Spilogale gracilis amphiala*) populations to increase following reduced competition from foxes. The SIA also demonstrated the low level of marine input from seabirds to the eagles' diets and their concentration, in particular, on introduced terrestrial prey.

Isotope studies are particularly useful for determining the effects of introduced fish, possibly because other techniques used for terrestrial vertebrates are often not applicable to aquatic species. SIA studies revealed how introduced salmonid species such as *Oncorhynchus* spp., *Salmo* spp., and *Salvelinus* spp. altered food webs by reducing prey fish abundance. This led to increased consumption of zooplankton by the invasive fish, and so to a reduction in their own trophic level (e.g., Vander Zanden *et al.* 1999). Introduced trout can also affect terrestrial food webs by consuming insects that would otherwise constitute important prey resources for terrestrial species. For example, trout introduced into previously fish-free lakes competed with the critically endangered mountain yellow-legged frog (*Rana muscosa*) for emergent insect prey (Finlay and Vredenburg 2007). Adult frogs feed on lake shores, and their consumption of emergent insects plays a key role in transferring energy between aquatic and terrestrial environments. Differences in isotopic values between benthic and pelagic prey revealed how fish altered aquatic food webs by consuming large numbers of benthic insects, largely restricting the supply of these to terrestrial environments. The importance and frequency of energy transfer between aquatic and terrestrial systems is increasingly recognised (e.g., Knight *et al.* 2005), and SIA is an ideal tool for examining such linkages.

Differences in niche width

Comparisons of niche width at individual and population levels can be explored with stable isotopes (e.g., Bearhop *et al.* 2006) and then used to predict the potential spread and range of an invader. Invasive species often show high plasticity of niche width in terms of habitat use, feeding ecology or behaviour (Hayes and Barry 2007). For example, a recent study in southern Sweden demonstrated that invasive signal crayfish (*Pacifastacus leniusculus*) have a potential niche width almost twice the size of the native European crayfish (*Astacus astacus*). However, signal crayfish often used a similarly sized niche to European crayfish within a given habitat (Olsson *et al.* 2009). Isotopic analyses also revealed greater plasticity in invasive plants. The invasive tree *Schinus terebinthifolius* in Hawaii had $\delta^{13}C$ values indicating a much greater capacity to adjust its physiology to variation in soil water availability, and more efficient water conservation, than the native trees to which it was compared (Stratton and Goldstein 2001).

Differences between multiple invasive species have also been examined with SIA, revealing how distinctions in habitat and diet utilisation allow multiple invasions of an ecosystem. Rudnick and Resh (2005) demonstrated that while Chinese mitten crabs (*Eriocheir sinensis*) and red swamp crayfish (*Procambarus clarkii*) primarily consumed plant material, crabs principally fed on aquatic algae, whereas crayfish consumed terrestrially-derived material. Likewise, Harper (2006) demonstrated how three invasive species of rats (ship rat, Pacific rats (*R. exulans*) and brown rat) on Pearl Island, New Zealand varied in their food resources and in their competitive ability to use them, allowing all three species to coexist on this small island.

Assessing priorities

Lastly, SIA of diets can aid the prioritisation of eradications. Once Roemer *et al.* (2002) demonstrated the threat that golden eagles posed to Channel Island foxes, eagles were translocated elsewhere and the best method for pig removal was implemented to avoid endangering the remaining foxes (Caut *et al.* 2006). In contrast, a combination of SIA, gut analyses and trapping led Quillfeldt *et al.* (2008) to conclude that a large breeding population of thin-billed prions (*Pachyptila belcheri*) on New Island, Falkland Islands, was not significantly impacted by invasive mice, ship rats and feral cats. Thus other islands within the archipelago could be prioritised for eradication programmes ahead of New Island.

INFORMING INVASIVE SPECIES MANAGEMENT

In addition to determining the effects of invaders, SIA can also help with formulating a response to invasions. By understanding the food used and locations from which it has been obtained, behavioural patterns can be identified that enhance the chances of successfully eliminating a population. An example of this approach was proved for invasive American mink (*Neovison vison*) in the Outer Hebrides, UK (Bodey *et al.* 2010). Stable isotopes can also help shape eradication protocols alongside a suite of standard techniques. For example, the likely outcomes of species eradication such as the disruption of trophic interactions, or the ecological release of other species, can be assessed more thoroughly prior to any eradication attempt as was carried out on a whole island basis prior to ship rat removal from Surprise Island, New Caledonia (Caut *et al.* 2009).

The logistics of eradicating common invasive mammals from small islands are now well understood. Successful eradications have continued to increase in size and complexity (Towns and Broome 2003: Veitch *et al*. 2011). However, increases in scale are accompanied by increased risks, including a higher risk of failure from unexpected challenges. While appropriate prior planning is of course essential, an adaptive management approach (Park 2004), which seeks improvements as progress continues, is often the most effective method for tackling these risks. The extent to which detailed knowledge about a species' ecology is required before an eradication campaign begins is debatable, particularly as the time taken to fully explore such questions may distract effort away from an efficient eradication campaign (Simberloff 2003).

Bodey *et al.* (2010) used SIA of diets of captured American mink early in an eradication attempt in order to reveal temporal patterns in mink behaviour that might assist the campaign. This approach identified that precise knowledge about what prey mink were consuming or when it was consumed was not necessary. Instead, the most useful information for the campaign was whether prey was marine or terrestrial in origin, and the relative time of consumption, coupled with background information on prey distributions. Individual variation in whisker and liver samples were used to generate a simple dietary time series. This revealed the continual importance of marine food sources to the population as a whole while the eradication progressed. Intra-sexual and intra-island differences were also found, and this again demonstrated that combining SIA with knowledge of prey distributions and gut analysis was crucial. For example, the presence at one locality of an additional terrestrial prey item, the introduced field vole (*Microtus agrestis*), is likely to have contributed to different behavioural patterns.

The use of SIA to inform eradication campaigns could greatly benefit invasive species management by revealing focal areas of foraging, habitats or areas in which animals may concentrate their time, and plasticity of responses to different trophic webs. The technique can help to refine methodologies and protocols, highlight areas for trap placement and assist with focussing of resources, potentially creating ecological traps for the target species.

CAVEATS AND CONCLUSIONS

Stable isotope analyses have transformed our understanding of numerous ecological questions about native species. There is a natural progression from these to its use for quantifying and resolving the effects of invasive species, and then to inform eradication campaigns. Unsurprisingly though, the use of SIA comes with some caveats. For example, there may be difficulties with interpreting the origins of food from multiple sources, variation in assimilation rates of isotopes into tissues both between individuals and species, and resolving the output of complicated statistical models (Crawford *et al.* 2008; Inger *et al.* 2008; Kurle 2009). Furthermore, additional work is required if we are to advance our understanding of turnover rates and growth times of specific animal tissues. On the other hand, while such information may be interesting, it may be beyond the information needed for an eradication programme. Thus, the few complexities with interpretation are not sufficient to prevent the incorporation of stable isotopes into management programmes. Given the value of adaptive management for the control or eradication of invasive species, SIA provides an additional tool with considerable potential to inform management options.

When combined with other methods, SIA can maximise the information obtained from culled individuals, enable rapid data accumulation, and thus inform areas of uncertainty as quickly as possible. Furthermore, it can aid preliminary studies on the feasibility of eradications, inform operations as they progress, and inform models of potential outcomes. Additionally, collection of samples such as fur, feathers and blood for SIA can be through non-lethal means, enabling its use to measure behavioural and dietary changes in endangered species pre- and post-eradication. Stable isotopes may also shed light on subsequent restoration attempts. For example, Gratton and Denno (2006) used changes in stable isotope values to show how disrupted trophic interactions were reconstituted after removal of the invasive *Phragmites australis* and restoration of *Spartina alterniflora* in a coastal saltmarsh. Stable isotopes can efficiently provide information at both the population and individual level from relatively small samples on species that may otherwise be difficult to study. They can be used to examine behavioural and ecological changes, and to describe the dietary and habitat plasticity of invasive species. They thus have great potential to inform management options, and should be seen as a powerful addition to the eradication toolkit.

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Heli-baiting using low concentration fipronil to control invasive yellow crazy ant supercolonies on Christmas Island, Indian Ocean

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Abstract Yellow crazy ants (*Anoplolepis gracilipes*) invaded Christmas Island sometime before 1935. By 2001, the species had formed destructive supercolonies over 2500 ha, or almost 30% of the island's rainforest. A heli-baiting operation in 2002 used high concentration fipronil (at 0.1 g/kg at 4 kg/ha) to eradicate all targeted supercolonies. However, supercolonies began to steadily redevelop across the island. We conducted surveys over the entire island from May to September 2009 and located 74 separate supercolonies that covered 833 ha. The boundary of each supercolony was mapped precisely by ground truthing. Two thirds of this area was too inaccessible and dangerous to be baited using standard hand-baiting techniques. Thus, in September 2009 we heli-baited 785 ha of supercolonies (with the remaining 48 ha intentionally not baited), using 3294 kg of ant bait, but this time using one tenth of the previous concentration of fipronil (0.01 g/kg at 4 kg/ha). All targeted supercolonies were again controlled, with ant activity reduced by 98.4% four weeks after baiting, and remained reduced by 99.4% 20 weeks after baiting. Direct non-target impacts of the baiting were minimal.

Keywords: Bait efficacy; surveys; non-target effects; land crabs

INTRODUCTION

Yellow crazy ants (*Anoplolepis gracilipes*) are one of the world's worst invasive species (Lowe *et al.* 2000) and are now widely distributed throughout the tropics (Wetterer 2005). These ants were accidentally introduced to Christmas Island some time prior to 1934 (Donisthorpe 1935). Ant numbers remained extremely low and had no obvious effects on the island's biota for decades. However, like many other invasive species of ants (Suarez *et al*. 2001; Holway *et al*. 2002; Tsutsui and Suarez 2003), crazy ants can form unicolonial (multi-queened) supercolonies where extremely high numbers of ants forage on the ground and in the canopy of rainforest trees (e.g., Haines and Haines 1978; Feare 1999). On Christmas Island, yellow crazy ants appear to benefit from a mutualistic relationship with introduced sap-sucking scale insects (Coccidae and Kerriidae) that secrete abundant, energy-rich honeydew (Abbott and Green 2007). As a consequence of this mutualism, the density of foraging yellow crazy ants within supercolonies typically exceeds 2000 ants per m^2 (or 20 million ants per ha) on the forest floor alone with 10.5 nests per m^2, which is the highest recorded density of foraging ants (Abbott 2005).

The first supercolony on Christmas Island was discovered in 1989 near the island's urban area, "the Settlement", where about 2 ha of forest were infested with crazy ants. No increases in the abundance of supercolonies were reported until 1996 (O'Dowd *et al.* 1999), following which untreated supercolonies expanded around their entire perimeter at rates of ~0.5 m per day (Abbott 2006). By December 1998, the total known infestation approached 200 ha, comprising 2-3% of the rainforest on Christmas Island (O'Dowd *et al.* 1999). Within four years, crazy ant supercolonies expanded to cover approximately 2500 ha, or more than 28% of the remaining forest. At supercolony densities, yellow crazy ants cause a rapid catastrophic shift in the rainforest ecosystems of Christmas Island, particularly through their impact on the red land crab (*Gecarcoidea natalis*) (O'Dowd *et al.* 1999, 2003; O'Dowd and Green 2009; Smith *et al.* subm.; see also Davis *et al.* 2008, 2010). Controlling infestations of yellow crazy ants on Christmas Island is of utmost importance for Christmas Island biota (Commonwealth of Australia 2006a, 2006b). This evolving crisis prompted an emergency response from the Australian Government (Green *et al.* 2004; Green and O'Dowd 2009). In September 2002, fishmeal baits with an active constituent of fipronil at 0.1g/kg were spread by helicopter (heli-baiting) at 4 kg/ha over 2509 ha of supercolonies. The campaign reduced ant abundance by an average of 99.4% within four weeks at all treated supercolonies (Green *et al.* 2004; Green and O'Dowd 2009).

Supercolonies again began to develop steadily across the island despite Christmas Island National Park (CINP) field teams' hand baiting 210 ha of supercolonies per annum with fipronil. The hand baiting did not keep pace with the rate of supercolony formation, particularly on the many inaccessible cliffs. By September 2009, over 800 ha of supercolonies again existed across Christmas Island (CINP unpubl. data).

Previous efforts to control or eliminate crazy ant supercolonies relied upon a relatively high concentration of fipronil. Here we document the efficacy of a 2009 heli-baiting campaign, which is the first crazy ant control programme to use low concentration fipronil (0.01 g/kg at 4 kg/ha) over a broad area.

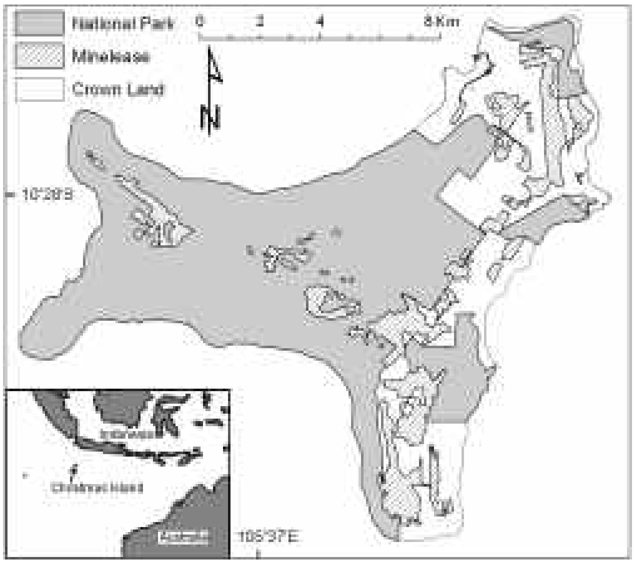

Fig. 1 Location and land tenure on Christmas Island.

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METHODS

Location

Christmas Island (10°25'S and 105°40'E) is an isolated oceanic limestone island of 135 km² in the north-eastern Indian Ocean 360 km south of Java and 2800 km west of Darwin. About 74% of the island is covered with natural vegetation comprised mostly of structurally simple, broad-leaved rainforest; 63% of the island comprises Christmas Island National Park (Fig. 1). The highest point is 361 m above sea level (Commonwealth of Australia 2006b). Christmas Island has a wet season from December to April, although rain may fall in any month of the year. Mean annual rainfall is 2068 mm, mean maximum temperature is 27.3° and the mean minimum temperature is 22.8° (Australian Bureau of Meteorology; Claussen 2005).

Field methods

Commencing in 2001, Christmas Island National Park conducted biennial surveys for yellow crazy ants, red crabs and other key biota at 877-1024 survey points (Fig. 2) spaced ~365.7 m apart across the entire island (Smith *et al.* subm.). Surveys were conducted during the dry season between May and September. At each survey point, teams of two field staff used two methods (one objective, one subjective) to assess whether that survey point fell within a supercolony. For a rapid, objective assessment of ant abundance, a 50 m transect was placed along the same bearing each year. These bearings were originally chosen randomly, although some were varied if extreme terrain made the site inaccessible. Each transect consisted of eleven sampling points located at 5 m intervals. At each sampling point, leaf litter was cleared with a swipe of the boot, and a laminated 20 x 20 cm card with lines dividing the card into four 10 x 10 cm quarters was placed on the cleared ground. One 10 x 10 cm quarter was selected at random. Observers then waited for 15 seconds before counting the number of ants that crossed the selected quarter over the ensuing 30 second period (cf. Abbott 2004; Green *et al.* 2004). Counting stopped if numbers exceeded 100 ants per 30 seconds. Counts were summed across the 11 card counts on each transect. Ant counts exceeding 37 ants per transect were identified as potential supercolonies because at these densities the ants tend to eliminate red crabs (CINP unpubl. data).

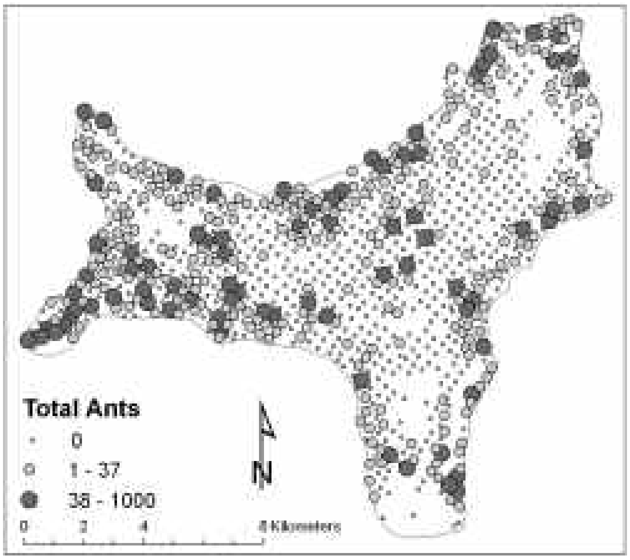

Fig. 2 Yellow crazy ant activity at 902 waypoints from the 2009 island-wide survey of Christmas Island. Black dots indicate waypoints with no ants recorded on ant activity cards; grey-centred circles indicate crazy ants were present but not in supercolony densities (1-37 ants on activity cards); large dark dots indicate that crazy ants were present at potential supercolony densities (>37 ants on activity cards).

At each survey point, and in transit between survey points, field teams also made subjective assessments of whether the area appeared to be a supercolony by looking for characteristic signs of crazy ant infestation including: 1) high crazy ant abundance on the ground and as 'trunk traffic' on trees; 2) large numbers of ant nests, typically at the base of trees and in rotten logs; 3) ant-infested red crab burrows; 4) dead red crabs (or other dead land crabs); 5) relatively large amounts of leaf litter; 6) relatively high numbers of scale insects; 7) excessive sooty mould; 8) giant African land snails; 9) relatively high numbers of seedlings; and 10) a relatively low diversity of 'other invertebrates', particularly 'other ants'.

The locations of any potential supercolonies discovered in transit between any of the survey points were recorded on hand-held GPS units (Garmin GPSmap 60CSx).

Following the objective and subjective assessments, each waypoint was then categorised as: 1) ants absent; 2) ants in low density; or 3) ants in a potential supercolony (on the basis of *either* of the objective or subjective assessment methods). These data were then used to generate a distribution map of potential supercolonies (Fig. 2) via ArcGIS 9.3.2.

Each potential supercolony was then revisited by field teams who mapped the precise location of its boundaries as follows. Three people walked 5-20 m apart along the length of the boundary with one person 'inside' the supercolony boundary continually searching for and confirming the *presence* of high densities of ants (and the supercolony characteristics listed above); one person 'outside' the supercolony boundary continually searched for and confirmed the *absence* of high numbers of ants; while the third person held the middle ground between the other two searchers. Through constant communication, the two outer people kept the middle person accurately positioned on the supercolony boundary. The person in the middle marked the boundary coordinates every 10-30 m using GPS and a hip-chain stringline to define a biodegradable cotton marker boundary to the supercolony. Most boundaries are easily identifiable by field crews on the ground. Occasionally, however, there was a wide 'transition zone' (cf. Abbott 2006) between heavily infested forest with high densities of ants and no live red crabs before reaching intact forest with very few or no crazy ants and many live red crabs. Although delineating the boundary required subjective assessment (particularly colonies with wide transition zones), the effectiveness and accuracy of this technique was regularly demonstrated as field crews – regardless of the size and complexity of the supercolony – always returned to within metres of the starting point.

Because of the fluid nature of supercolony boundaries, their perimeter needed to be delineated as soon as possible before the actual heli-baiting. Thus, boundary marking began on 4 August 2009 and continued until 16 September 2009, with the last of the supercolony boundaries being delineated while heli-baiting was under way elsewhere (see below). This methodology produced up-to-date detailed maps of every crazy ant supercolony on the island, with very finely resolved boundaries (Fig. 3).

Heli-baiting

AntOff ant bait, with the active ingredient fipronil at 0.01 g/kg, was supplied by Animal Control Technologies

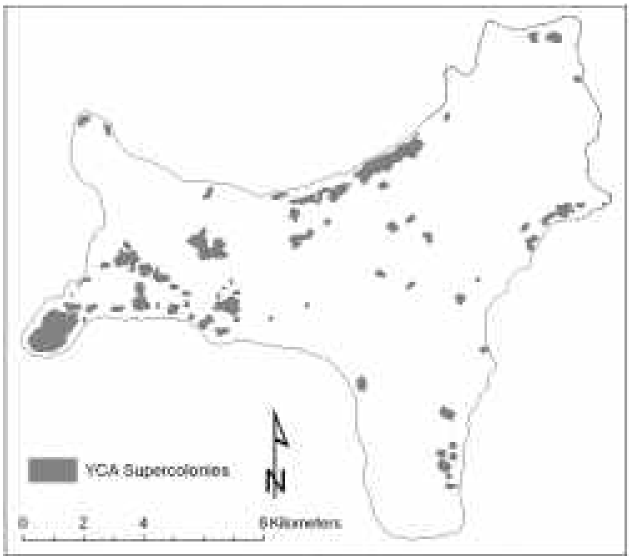

Fig. 3 Distribution of yellow crazy ant supercolonies following boundary ground-truthing by field teams prior to heli-baiting in September 2009.

(Australia) Pty Ltd in the form of small pellets, roughly 1.5 x 1.5 mm in cross section, and between 2 and 6 mm long. The 7000 kg of bait was packaged in 12.5 kg plastic-lined cardboard boxes and transported to Christmas Island by ship.

Heli-baiting was planned for September 2009, the end of the dry season. This month was chosen because: 1) bait delivery and bait uptake by the crazy ants would be impeded during wet weather; 2) land crab activity is minimal at the end of the dry season as red crabs tend to remain in their burrows, thereby reducing the potential for non-target contact with the bait; and 3) the rainforest canopy is at its most open enabling more bait to fall to the forest floor (cf. Green *et al*. 2004).

AntOff baits were dispersed over supercolonies from a Bell 47 Soloy helicopter operated by McDermott Aviation Pty Ltd. The bait delivery mechanism used was developed by McDermott Aviation for the 2002 heli-baiting operation on Christmas Island and described by Green *et al*. (2004) and Green and O'Dowd (2009). Essentially, bait was dispersed from an inverted conical bucket suspended below the helicopter. Bait flowed through a 25 mm diameter aperture in a base plate at the bottom of the bait bucket and onto a rotating spreader powered by a petrol-driven, four-stroke engine attached to the framework of the bucket. Pilots entering the air space above a supercolony boundary electronically opened a sliding gate beneath the aperture in the bucket, thereby enabling the bait to flow onto the spreader. This resulted in an even spread of baits for 12 m either side of the helicopter at a rate of roughly 4 kg per hectare when the pilot flew at 100 km per hour.

Supercolony boundaries were defined for the helicopter pilots on ArcMap layers. The pilot used a Trimble differential GPS unit with sub-metre accuracy to ensure that baits were spread to the edge of supercolonies and that flight paths were straight and the correct distance apart, which gave continuous and even spread of bait over the entire target supercolony.

Five supercolonies or subsections of supercolonies, each about 5 ha, were deliberately left unbaited for an ongoing research project into biocontrol of scale insects (a joint collaboration between the Director of National Parks, Christmas Island National Park, Monash University and La Trobe University). Three small supercolonies (9 ha total) on a steep slope near the township were baited by hand because the local community raised concerns about human safety. One supercolony and one subsection of a supercolony were not treated because they were intentionally set aside for an ongoing alternative baiting research project (2 ha). Because fipronil can have strong negative effects on freshwater fauna (e.g., Maul *et al*. 2008), we did not bait two supercolonies (12 ha) that were within 200 m of Ramsar Wetlands of International Importance (Hosnie's Spring and The Dales). In total, 48 ha were not treated during this heli-baiting campaign.

One non-target species susceptible to fipronil is the robber crab (*Birgus latro*), which is attracted to AntOff baits (CINP unpubl. data). In order to minimise robber crab mortality, we created food lures designed to entice them away from baited sites. In the weeks prior to heli-baiting, 4000 kg of chicken feed pellets was mixed with 320 kg of shrimp powder ('Belacan') in concrete mixers and placed into 12 kg bags. These bags were stored for as long as possible, which allowed the shrimp powder to infuse with the chicken food pellets. One or two days before heli-baiting, the helicopter was used to drop lure stations (3-4 kg of chicken feed / shrimp powder mixture) at intervals 50 m from the mapped supercolony boundaries. Lure stations were delivered from a different bucket slung beneath the helicopter to ensure that the chicken pellets were not contaminated with any residual fipronil. This method effectively lured most robber crabs out of areas to be baited (CINP unpubl. data). In total, 1105 robber crab lure stations were deposited from the helicopter.

Christmas Island National Park has engaged CESAR Consultants Pty Ltd., as independent consultants to quantify direct and indirect (bioaccumulation) impacts of baiting on non-target species. These data are still being collected.

Monitoring bait efficacy

The effects of aerial baiting on ant density were assessed at nine supercolonies (Table 1), which provided a range of densities and locations across the island. In addition, the four most accessible of the five untreated biocontrol research project plots were used as control sites to monitor the density of ants without chemical treatment (Table 1).

Estimates of ant densities in trial supercolonies were obtained using standard Christmas Island National Park methods employed since 2001: 3 x 50 m straight line transects were established within the boundary of each

Table 1 Mean pre-treatment ant densities and areas of monitored supercolonies.

Supercolony ID	Initial Ant Density	Area (ha)	Treatment
917	24	25.7	Baited
372	67	100.7	Baited
135	69	12.5	Baited
368	88	5.3	Baited
538	102	4.4	Baited
467	144	8.1	Baited
252	158	30.5	Baited
148	174	29.0	Baited
184	528	63.1	Baited
403	182	5.4	Control
582	200	4.8	Control
318	238	5.2	Control
206	414	4.8	Control

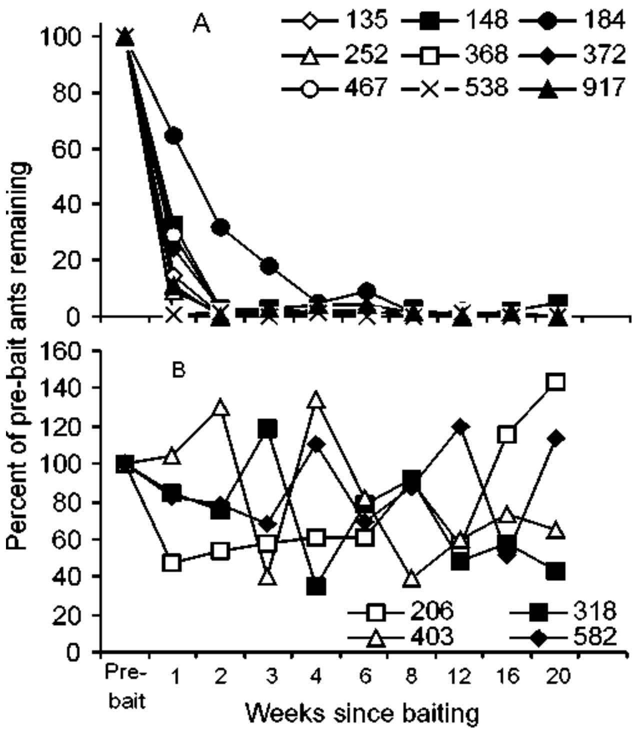

Fig. 4 Ant activity at **(A)** nine baited and **(B)** four unbaited (control) supercolonies expressed as a percentage of the of mean ant activity for three weeks prior to baiting for each supercolony. Numbers in the legend indicate supercolony identity.

supercolony; each transect was located at least 50 m from a boundary, and at least 50 m from a neighbouring transect; and each transect consisted of eleven survey points at 5 m intervals marked with flagging tape. At each survey point, ant cards were used to estimate ant activity per 30 seconds using identical methods described above for the island-wide survey of ants. Counts were summed across the 11 card counts on each transect. Ant activity was defined as the mean of ant counts from the three transects within each supercolony. Ant counts were conducted weekly for three weeks prior to baiting (to obtain a pre-baiting mean density), and then 1, 2, 3, 4, 6, 8, 12, 16 and 20 weeks after baiting.

RESULTS

Surveys for crazy ants over Christmas Island revealed that 542 of the 902 (60.1%) waypoints had ant infestations with potential to become supercolonies (Fig. 2). Ground truthing of all potential supercolonies and those discovered in transit revealed 74 discrete supercolonies covering 833 ha (Fig. 3), with supercolony area between 0.4 and 141.1 ha (mean 11.5 ha). Forty-two supercolonies covering 511.9 ha (or 65.2% of total heli-baited area) were in areas too rugged to bait by hand.

Baits were spread from 4-19 September 2009 covering all 784.8 ha of targeted supercolonies with 3294 kg of bait at a mean application rate of 4.2 kg per ha. GPS downloads revealed remarkably few inaccuracies during the aerial baiting campaign, with no baits being spread outside the targeted areas.

Ant densities declined by a mean of 79.3% (±20.1 SD) one week after baiting and 98.4% (±1.9 SD) four weeks after baiting (Fig. 4a). This reduction was sustained, with ant numbers reduced by 99.4% (±1.6 SD) some 20 weeks after baiting (Fig. 4a), when 288 of 297 (97.0%) ant sampling points still had zero ants per 30 seconds on ant count cards. Ant activity in control plots remained high, although it varied over the monitoring period (Fig. 4b).

Within baited supercolonies, the percentage decline in ant activity one week after treatment was negatively correlated with log pre-baiting ant activity (linear regression $F_{1,9}$=6.4; P=0.04); low density crazy ant supercolonies declined more rapidly than high density supercolonies. For example supercolony 184 initially had an average of 528 ants per transect and declined more slowly than supercolony 368, which initially had 86 ants per transect (Fig. 4).

DISCUSSION

The yellow crazy ant heli-baiting campaign on Christmas Island in 2009 was a complete success. The entire island was surveyed for ants, all supercolonies were delineated, all targeted supercolonies were heli-baited on time, and, importantly, all monitored supercolonies showed decreases in ant activity to well below supercolony level. Within four weeks of baiting, virtually no crazy ants were recorded on ant activity cards within baited supercolonies, and this pattern has continued for the first 20 weeks after baiting.

This is the first attempt to control yellow crazy ants on a broad scale using fipronil at 0.01 g/kg at 4 kg/ha. For example, in Arnhem Land, northern Australia, yellow crazy ant supercolonies are treated with 0.01 g/kg at 10 kg/ha (B. Hoffman pers. comm.). Between 2000 and 2009, Christmas Island National Park used fipronil at 0.1 g/kg at between 4 kg /ha (e.g., Green *et al.* 2004; Green and O'Dowd 2009) and 6 kg / ha (CINP unpubl. data). These higher doses were understandable given the urgency and novelty of the yellow crazy ant situation in 2001, where almost 30% of the island had become heavily infested with crazy ant supercolonies (Green and O'Dowd 2009), and failure to control the supercolonies would have been disastrous for the Christmas Island biota. In the 2002 heli-baiting campaign, Christmas Island National Park achieved a 99.4% knockdown of yellow crazy ants in all monitored supercolonies (Green and O'Dowd 2009). We achieved an identical knockdown (99.4%) using a ten-fold lower concentration of active ingredient. In total, 31 g of fipronil was used to eradicate 785 ha of supercolonies.

It may be possible to further reduce the concentration of fipronil used to control supercolonies, particularly those with less dense ant populations. For example, supercolony 184 had the highest density of ants recorded on the island. Despite the lower concentration of fipronil used in this programme, this supercolony was eradicated within four weeks.

Christmas Island National Park has been conducting chemical baiting trials since 2000 to determine the most effective method of controlling yellow crazy ants (CINP unpubl. data). Despite trialling hydramethylnon, pyriproxyfen and indoxacarb, fipronil has proven to be the only effective option for controlling yellow crazy ants on the island. Surprisingly, hydramethylnon effectively eliminated yellow crazy ant supercolonies in Arnhem Land (B. Hoffmann pers. comm.).

Fipronil is a phenylpyrazole broad spectrum insecticide effective at low field application rates against a wide range of arthropods (including crustaceans), even those often resistant to other insecticides, such as pyrethroids, organophosphates and carbamates (Narahashi *et al.* 2007).

However, it is unlikely that the heli-baiting campaign on Christmas Island heavily affected non-target species for several reasons. First, Christmas Island National Park only treats high density ant infestations (i.e., supercolonies). In these areas, non-target impacts are minimal since most native invertebrates have already been killed by the crazy ants. Furthermore, crazy ant activity is so high in such areas they remove bait at rates of 7% per minute (Marr 2003), which limits bait exposure to surviving native species. Christmas Island National Park can not apply fipronil to areas containing crazy ants at low densities because the non-target impacts would be catastrophic.

One native invertebrate that can enter baited supercolonies is the large (up to 6 kg), nomadic robber crab. This species usually survives for some time as it passes through a crazy ant supercolony but is also highly susceptible to fipronil poisoning. We used lure stations around selected supercolonies to attract robber crabs and found more than 100 individuals at one lure station within 24 hours of placement. There was low mortality of crabs around baited supercolonies even where crabs were known to be adundant nearby (CINP unpubl. data). Further, no red crabs were found dead within or around baited supercolonies. Either the red crabs were not sufficiently attracted to the AntOff bait to emerge from their burrows during the heli-baiting campaign or the yellow crazy ants monopolised baits before red crabs from outside the supercolony could locate them.

Data collected during the 2002 heli-baiting campaign indicated that most of the aerially-delivered ant bait successfully passed through to the forest floor. If bait remained within the forest canopy, it was most likely to be consumed by crazy ants (Green and O'Dowd 2009). There was no evidence of an impact of fiprinol on native canopy arthropods, arboreal geckoes or land birds (Stork *et al.* 2003), nor was there any evidence of impacts on native leaf litter invertebrates (Marr 2003). There was no residual fipronil detected in the soil one week, one year or two years after aerial baiting in 2002 (Marr 2003). Given that we used fipronil at a lower concentration, we expected even fewer non-target impacts from the 2009 heli-baiting campaign.

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The Quail Island story – thirteen years of multi-species pest control: successes, failures and lessons learnt

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Abstract. Quail Island (Ōtamahua) is an 85 ha island in Lyttelton Harbour, Banks Peninsula, New Zealand. Since 1997, community volunteers have eradicated rabbits (*Oryctolagus cuniculus*), cats (*Felis catus*), hedgehogs (*Erinaceus europaeus*), and ship rats (*Rattus rattus*) from the island as preliminary steps towards ecological restoration. At present, a network of traps on the adjacent mainland and a stepping-stone island successfully intercepts mustelids and other unwanted vertebrate pests en route to Quail Island. However, the public use of the island, its close proximity to, and inter-tidal link with, the mainland makes this island a significant risk to reinvasion, particularly by rodents. Lessons learnt from 13 years of pest work are outlined.

Keywords: Rodents, mouse, ship rat, hedgehog, mustelids, stoat, eradications, *Erinaceus europaeus*, *Rattus rattus*, *Mus musculus*, *Mustela erminea*, brodifacoum, ecological restoration

INTRODUCTION

Quail Island (Ōtamahua) is an 85 ha Recreation Reserve administered by the Department of Conservation, located in Lyttelton Harbour (43° 38′ S, 172° 42′ E), Canterbury, New Zealand (Fig. 1). The island is dominated by improved exotic grasslands, including cocksfoot (*Dactylis glomerata*), browntop (*Agrostis capillaris*), Yorkshire fog (*Holcus lanatus*) and several *Bromus* species (Burrows *et al*. 1999), with areas of native restorative planting across the island.

Quail Island is considered a 'mainland island' rather than a true island, as exposed mudflats provide a land bridge at low tide from Moepuku Point on the mainland via King Billy Island to Quail Island (Fig. 1). Consequently, the island is vulnerable to invasion by mammal pest species. It is unknown whether the introductions of these pests were deliberate or accidental.

In 1997, the New Zealand Department of Conservation, representatives of local Maori Te Rūnanga o Ngāti Wheke and dedicated volunteers began provisional planning for the ecological restoration of Quail Island (Burrows and Leckie 2001; Bowie *et al*. 2003; Norton *et al.* 2004; Bowie 2008). A major impediment to the restoration process was the presence of mammalian predators and the potential for ongoing reinvasion across the land bridge.

In this paper we describe a programme to eradicate mammalian pests from Quail Island (see Fig. 2) and the on-going control of reinvading mustelids (stoats, *Mustela erminea*; ferrets, *M. furo*; and weasels, *M. nivalis vulgaris*), hedgehogs (*Erinaceus europaeus*) and feral cats (*Felis catus*). The experience and knowledge we have gained over the 13 years since this multi-species pest eradication programme began are also discussed. As the pest control work was carried out by volunteers, robust scientific design was not a high priority; however, sufficient planning was carried out and records taken to ensure lessons could be learnt throughout the programme. We believe other groups undertaking future eradication operations such as those attempted on Quail Island could benefit from our experiences.

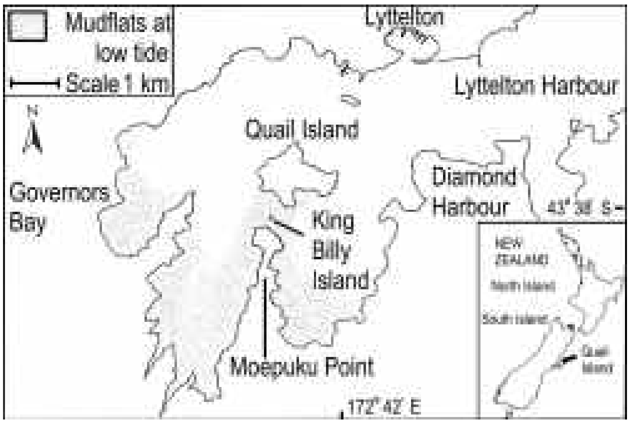

Fig. 1 Lyttelton Harbour showing Quail Island, King Billy Island and Moepuku Point.

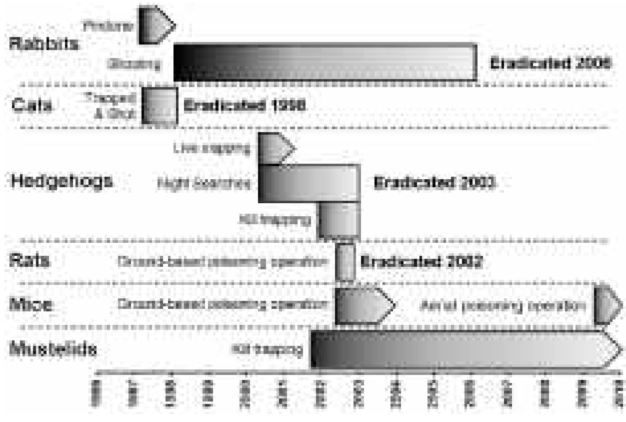

Fig. 2 Timeline of eradications undertaken on Quail Island and King Billy Island between 1997 and 2010.

METHODS AND RESULTS

Possum, rabbit and cat eradication

Possums (*Trichosurus vulpecula*) were eradicated from Quail Island in 1988, before the current project began (Brown 1999).

In 1997, pindone cereal bait (0.25g/kg pindone) was aerially applied twice to reduce the existing rabbit (*Oryctolagus cuniculus*) population (Brown 1999; Burrows and Leckie 2001), with remaining survivors shot or accidentally trapped. The last known rabbit on Quail Island was a male caught in a Fenn trap (Mk 6) (FHT Works, Redditch, England) set for mustelids in 2006.

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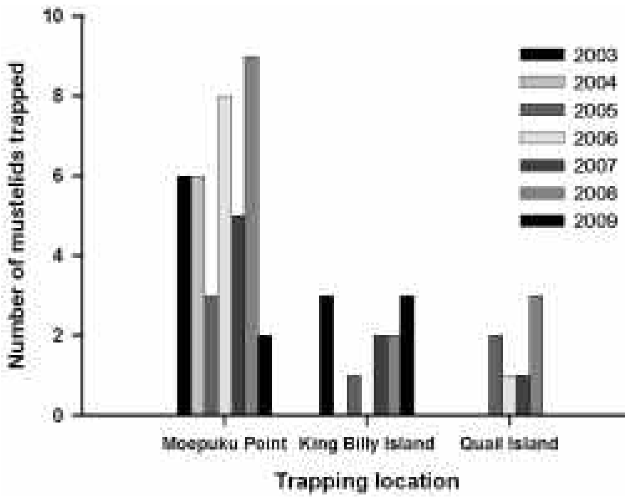

Fig. 3 Number of mustelids trapped on Quail Island, stepping-stone King Billy Island and adjacent mainland site Moepuku Point between 2003 and 2009.

The last feral cat was removed following a shooting and Fenn trapping regime in 1998. In total, 10 cats were removed from Quail Island at a cost of 68 worker hours. Since 2003, seven cats have been captured on Moepuku Point; however, none have been sighted or captured on King Billy Island or Quail Island.

Mustelid control

In 2001, wooden trap boxes, each containing two Mk 6 Fenn traps, were set up in a 120 m × 120 m grid over Quail Island (n = 62) and King Billy Island (n = 6). Traps were set primarily for mustelids and baited with hen eggs. It was anticipated that hedgehogs and rats would also be caught as by-catch. In 2002, an additional six trap boxes were placed on the northern-most tip of Moepuku Point, the closest mainland site to Quail Island, to intercept invading predators. All trap boxes were labelled and GPS coordinates recorded for monitoring purposes. Traps were repositioned or concentrated, depending on relative catch success. Detailed methods are described in Kavermann *et al.* (2003).

Analysis of data collected show that 39 mustelids were trapped on Moepuku Point between 2003 and 2009 (21 ferrets, 16 stoats, and two weasels). During the same period nine stoats and three weasels were trapped on King Billy Island, while four stoats, one weasel, and one possible ferret were caught on Quail Island (Fig. 3). Cats, hedgehogs and rats were also trapped on Moepuku Point (Table 1).

Table 1 Comparison of pest species trapped on Quail Island, stepping-stone King Billy Island and adjacent mainland site Moepuku Point between 2003 and 2009.

Pest Species	Moepuku Point	King Billy Island	Quail Island
Ferret	21	0	1#
Stoat	16	9	4
Weasel	2	3	1
Hedgehog	2	0	1*
Cat	7	0	0
Rat	11	8	0

#The animal was in a poor state for identification and we suspect it was misidentified as none have been trapped on King Billy Island.
*Evidence from tracking tunnels and scats suggest this animal was a survivor from the original Quail Island population and not a recent immigrant from the mainland.

Hedgehog eradication

Live trapping for hedgehogs was conducted for 11 consecutive nights in January 2000, using wire cage traps and wooden treadle traps baited with dog roll. Traps were set 150 m apart near tracks and fence lines and checked daily (see Thomsen *et al.* 2000 for detailed methods). Hedgehogs were also trapped in the Fenn traps set for mustelids.

Spot-lighting for hedgehogs took place on 13 occasions since 2000. Searchers wearing headlamps walked tracks between 17:30 hrs and midnight, collecting hedgehogs and recording their location. Live-captured hedgehogs were humanely euthanased and stomach contents stored for analysis (for details see Thomsen *et al.* 2000). The density of hedgehogs was estimated as 0.69/ha by dividing the total number caught by the size of Quail Island.

A total of 59 hedgehogs were removed from Quail Island between January 2000 and October 2003. The initial 11 nights of cage trapping removed 24 hedgehogs and represents an average of 2.2 captures/night. Spot-lighting over the first six nights of searching collected 23 hedgehogs or 3.8 captures/night. Fenn traps captured an additional 10 animals, including the last known hedgehog removed in 2003.

Rat eradication

In August 2002, the eradication of rats from Quail Island commenced with the establishment of 555 bait stations placed in a 40 m × 40 m grid over the island. A combination of yellow Pestoff bait stations (n = 351) and custom made Novacoil stations (450 mm lengths of 110 mm diameter black non-perforated plastic Novacoil drain pipe; n = 204) were used (see Kavermann *et al.* 2003 for further details). In December 2002, the dominant vegetation in a 20 m radius surrounding each bait station was recorded as grass, trees or scrubland, and this information was used to assess bait take in different habitats.

At the commencement of the operation, ten cereal Pestoff 20R rodent pellet baits (0.02 g/kg brodifacoum) were placed in each bait station. In the first seven days, stations were checked daily and bait replenished or increased to 20 pellets in cases where all bait had been removed. In the subsequent five weeks, all stations were checked every two days and bait replaced as required.

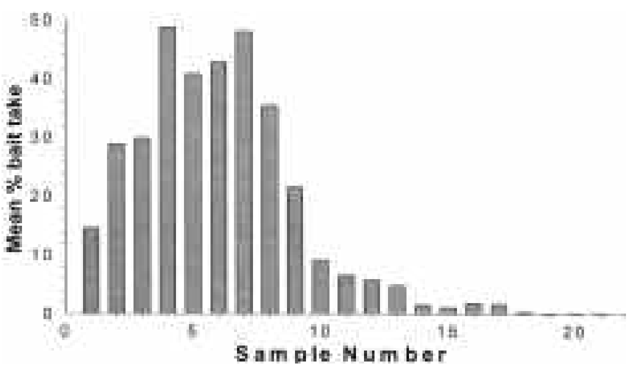

Fig. 4 Percentage bait take for all bait stations during the initial 21 samples (37 days).

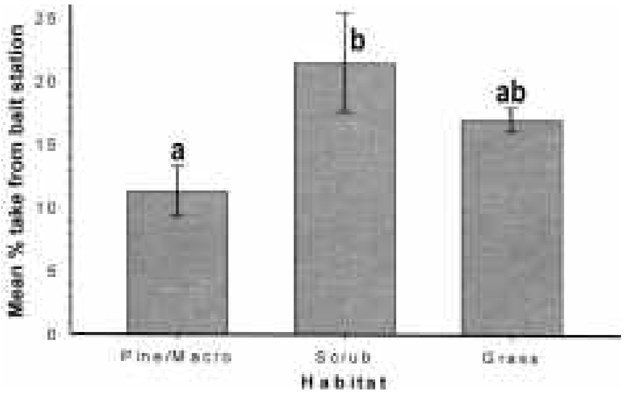

Fig. 5 Mean percent bait take (±SEM) in stations by rodents in three habitats on Quail Island. Differing letters above bars denote significance at 5% level of probability using LSD test.

In late September 2002, when bait take ceased, a single 20 g Talon 50 WB wax impregnated cereal 'egg' (0.05 g/kg brodifacoum) and five 20R Pestoff baits were placed in each station to overcome any possible aversions to the original baits. The higher concentration of poison in the new bait also meant that a smaller amount was required for target animals to consume a lethal dose. After one week the cereal 'egg' baits were wrapped in tin foil to minimise the effects of slugs (*Deroceras* spp.), insects and decomposition due to moisture and returned to each bait station for any remaining rodents.

Bait take was used to assess rodent presence and activity during the baiting operation. Preliminary eradication was considered achieved when bait take stopped. Detailed descriptions of bait take calculations for rodents during the operations were provided by Kavermann *et al.* (2003).

Overall, percentage bait take from Novacoil bait stations was significantly higher ($F_{1,418}$ = 16.83, P <0.001) than from the Pestoff bait stations for the entire poisoning operation. A steady increase in bait take occurred in the first four days of the operation, peaking at 49% at sample 4 (day 4) and again at 48% at sample 7 (days 8 and 9). This was followed by a steady decrease in bait take with minimal interference after sample 18 (days 30 and 31) (Fig. 4). Rats were considered eradicated at day 38.

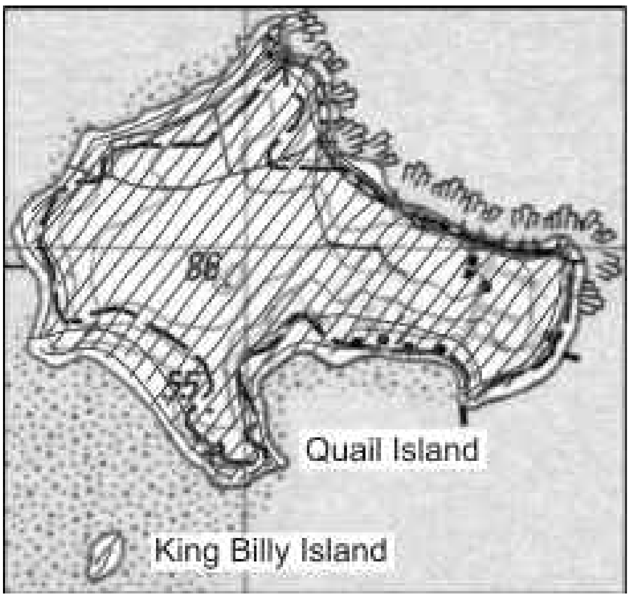

Fig. 6 GIS generated flight lines from first helicopter baiting on Quail Island and King Billy Island on 26 July 2009.

Percentage bait take on Quail Island was significantly ($F_{2,534}$ = 16.72, P <0.001) different between habitats. Pair-wise comparisons of means (LSD test; α = 0.05) indicated that bait take in scrubland was significantly higher than in mature pine (*Pinus* spp.) and macrocarpa (*Cupressus macrocarpa*) stands (Fig. 5).

Mouse eradication

Although rats were successfully eradicated using a bait station operation in 2002, mice were not. Subsequent aerial operations to eradicate mice were undertaken on Quail Island and King Billy Island on the 26 July and 6 August 2009. To ensure thorough bait coverage across cliff faces, the helicopter pilot baited the island by flying twice around the coast, and the then by flying in several northeast-southwest sweeps (Fig. 6). The second sowing (6 August 2009) was identical, except that flying was carried out in a northwest-southeast direction. The intended bait-sowing rate of 8 kg/ha was monitored on mown tracks using 50 m^2 transects. Where possible, tracks perpendicular to the flight line were used and included a selection of locations both coastal (six transects) and central (nine transects). The mean sowing rate around coastal areas was 8.2±0.8 kg/ha, while sowing rates in the island centre averaged only 3.2±0.3 kg/ha.

Pre- and post-eradication operation mouse populations were monitored following Gillies and Williams (2002), using 99 Black Trakka tracking tunnels (Connovation, Auckland Ltd) baited with peanut butter and placed in a 100 m grid over Quail Island. The tracking tunnels were placed out one week before the first drop, and repeated thereafter from one week after the first drop. Standard snap-back mouse traps baited with unheated popcorn and peanut butter were also placed in the centre of rat/mustelid trap boxes on King Billy Island and Quail Island as another monitoring tool. To determine whether subsequent mice found on Quail Island were new invaders or survivors of failed eradication, mouse tail tips were collected from the island prior to the poisoning and stored frozen in 100% ethanol as reference DNA for future molecular analysis (Dilks and Towns 2002; MacKay *et al*. 2007). Mouse activity was recorded on 83% of tracking cards one week before the first aerial poison drop but were eliminated a week after the last aerial poison drop. No signs of mice were recorded on either island for six months after the drop, until a mouse was caught in a trap box on King Billy Island on 23 February 2010.

DISCUSSION

Rabbits and cats

Only a few rabbits survived the initial Pindone poisoning operation, possibly going underground for a period of time (Brown 1999). The final few rabbits proved elusive and it was unexpectedly a Fenn trap set for mustelids that removed the final individual in 2006. The nine cats intercepted on Moepuku Point highlights the value of these traps for maintaining the integrity of Quail Island as a refuge for native species.

Mustelids

A large number of mustelids have been trapped on Moepuku Point, though few have made it to Quail Island. This demonstrates the importance of interception trapping on the mainland to reduce the threat of mammalian pests invading the island. However, traps are still needed on Quail Island to kill those animals that may reach it. Furthermore, ongoing monitoring of capture success is vital for reviewing trapping strategies and maximising trapping success.

Hedgehogs

The eradication of hedgehogs from Quail Island is the first reported success of its kind achieved on a New Zealand island. At an estimated density of 0.69 hedgehogs/ha, Quail Island hedgehog habitation was very low compared to other studies which show as many as 1.1-2.5 hedgehogs/ha (Brockie 1974). Most (93%) of the 59 hedgehogs were removed from the grassland areas, indicating it may be a preferred habitat. Night searches were particularly successful at track intersections and close to the stock dam, the only open body of water on Quail Island. Hedgehogs appeared to prefer the mown tracks for ease of movement and feeding, and were observed by searchers to feed on invertebrates, particularly slugs and slaters (*Porcellio scaber*). Brockie (1990) proposed that hedgehog densities reflected invertebrate food availability, a finding supported by Bowie (unpublished) who found invertebrates to be more abundant in exotic grasslands compared with other habitats on Quail Island. Grasslands also provide a greater abundance of skinks (*Oligosoma* spp.), another known food source of hedgehogs (Moss and Sanders 2001) and found in the stomachs of specimens taken from Quail Island (Kavermann *et al*. 2003). The absence of hedgehog scats on tracks and lack of prints from tracking tunnels since the last trapped individual (27 October 2003) suggest that hedgehogs have been successfully eradicated from the island.

Rats

The eradication of rats in 2002 was another successful operation, although mice were not similarly eradicated. We had anticipated challenges in successfully eradicating rodents from Quail Island because of the thick exotic grasses and the chances of rodents encountering bait stations. We therefore used 40 m spacing between bait stations, which was closer than other successful island bait station eradication operations for rats (eg. 50 m spacing used by Taylor and Thomas 1993). The greater success of the Novacoil bait stations may have been due to their wider entrance, making them easier to locate and access by rodents. Novacoil stations entrances were also positioned at ground level and did not require animals to step up into them, unlike the Pestoff stations. The significantly lower bait take from the Pestoff bait stations (Kavermann *et al*. 2003) would support this hypothesis. Recent work by Spurr *et al*. (2007) supports the view that entrance size is important for rats. Based on our results, we recommend the use of Novacoil stations or other similar-sized bait stations to increase the probability of rodents encountering more bait. Novacoil bait stations were also cheaper, more robust and the material is readily available.

Mice

Several factors may have contributed to the failure to eradicate mice using the bait stations. First, the 40 m bait station spacing was likely too wide for mice, as they have smaller home ranges than rats (Ruscoe and Murphy 2005). As such, all mice were unlikely to encounter at least one bait station, which jeopardises a key component of eradication in that every individual must be put at risk (MacKay *et al*. 2007). In contrast, several successful mouse eradications from islands have used station grid spacing of 25 m (Thomas and Taylor 2002). While the 40 m spacing was the likely cause of the failure, other factors may also have contributed. For example, during their study on Hawea Island, Taylor and Thomas (1989) noticed that large male rats defended bait stations from smaller rats, a behaviour also likely to deter mice. This dominant behaviour observed by Taylor and Thomas (1989) may also help to explain the prolonged bait take on Quail Island when compared with similar eradication attempts on other islands. After the dominant animals succumb to poison, the less dominant individuals (both rats and mice) can access the bait stations. It appears this may be the case with Quail Island as bait take continued for 37 days.

The use of aerially applied baits for eradication attempts of rodents on islands has historically given the best rate of success, particularly where cliffs make it difficult to use alternative control strategies (Howald *et al*. 2007). Aerially applied brodifacoum is the most widely used poison for mice on islands. Although this has a record of successful eradications, the overall success rate of mouse eradications on islands is only 49% (MacKay *et al*. 2007). Bait coverage, particularly where extensive exotic grassland is present, seems to be critically important for success. The lower bait coverage in the interior of Quail Island (3.2 kg/ha) may have allowed mice to survive. Also, the mixture of thick exotic grasses offers ground cover for mice to move through and may prevent them from encountering baits. Furthermore, grasses provide a good seed source for mice, therefore individuals may not require any supplementary food source encountered in baits.

Unfortunately the mouse caught on King Billy Island and a mouse track recorded on Quail Island suggests mice may have reinvaded Quail Island from the mainland. For future management of mice it is essential to know whether they are survivors from the aerial eradication attempt or recent invaders. DNA collected from the mice will hopefully provide this answer.

KEY LESSONS

1. Interception of mustelids and rats on Moepuku Point and King Billy Island is helping to reduce the number of invaders reaching Quail Island.

2. Monitoring trap catch locations with well labelled traps and keeping thorough records is essential for managing efficient reinvasion strategies so that traps can be repositioned or concentrated, depending on relative catch success.

3. A mixture of eradication methods for hedgehogs (eg. cage trapping and spot-lighting on tracks) is useful to initially reduce numbers, but kill traps may be most successful at lower population densities. Mowing tracks in exotic grassland may also be a strategy to allow more effective spot-lighting.

4. We recommend using unheated popcorn as an alternative mouse bait, as peanut butter is often also eaten by invertebrates.

5. Bait stations with larger entrances, such the 110 mm Novacoil, have better bait take than bait stations with smaller entrances, particularly in thick exotic grasses.

6. A bait station grid spacing of 40 m achieved the goal for eradicating rats.

7. Given molecular advances, keeping DNA from pest species being eradicated will be important to distinguish between new invaders and survivors of failed eradications.

8. The use of GPS on helicopters does not guarantee correct bait sowing rates and deposition on the ground. Transects should be used on open areas such as wide tracks to check how much bait is present on the ground to confirm adequate bait application.

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Earth, fire and water: applying novel techniques to eradicate the invasive plant, procumbent pearlwort *Sagina procumbens*, on Gough Island, a World Heritage Site in the South Atlantic

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Abstract The Eurasian plant procumbent pearlwort (*Sagina procumbens*) was first reported in 1998 on Gough Island, a cool-temperate island and World Heritage Site in the central South Atlantic. The first population was discovered adjacent to a meteorological station, which is its assumed point of arrival. Despite numerous eradication attempts, the species has spread along a few hundred metres of coastal cliff, but has not as yet been found in the island's sub-Antarctic-like mountainous interior. At South Africa's sub-Antarctic Prince Edward Islands *Sagina* is spreading rapidly in vegetated and unvegetated habitats, and is considered beyond control. A similar situation could eventuate on Gough Island if the plant spreads inland, with deleterious effects on the island's ecosystems. Eradication methods progressively used on Gough Island included mechanical removal and dumping of plants and seed-infested soil at sea well away from the island, application of herbicides to kill both growing plants and germinating seeds, gas flames to kill seeds and seedlings in rock cracks, near-boiling water to kill seeds in soil, high-pressure water jets to strip infested areas of soil and peat down to bedrock, and spraying with salt water. Germination trials have shown that spraying with sea water inhibits seedling production and a steady decline in seed load in infested areas over almost a decade. However, eradication has been hampered by the plant's inconspicuous nature, fast growth rate, large seed production leading to an equally large seed bank, long-lived seeds, difficult terrain that requires qualified rope-access technicians to work in safety, and the island's remote location. Although eradication has not yet been achieved, *S. procumbens* remains confined to its current restricted distribution on the island.

Keywords: Inconspicuous, long-lived, seed bank, mechanical removal, herbicide, salt water, pressure spray

INTRODUCTION

Gough Island, a cool-temperate oceanic island in the mid South Atlantic, has often been described as one of the most important seabird islands in the world (Ryan 2007). The island is part of the United Kingdom Overseas Territory of St Helena, Ascension and Tristan da Cunha. The 6400-ha island and its surrounding 12 nautical mile territorial waters have been designated a nature reserve under the Conservation of Native Organisms and Natural Habitats (Tristan da Cunha) Ordinance of 2006, as a World Heritage Natural Site since 1995 (expanded to include Inaccessible Island in the Tristan Group in 2004), and as a Ramsar Wetland of International Importance since 2008. The island has also been listed as an Important Bird Area and an Endemic Bird Area (Ryan 2008). Gough supports over 70 species of indigenous vascular plants (Ryan 2007), four of which are endemic to the island and a further 25 endemic to the Tristan da Cunha Group (Jones *et al*. 2003). Activities on the island are controlled through a management plan adopted in 1993 (Cooper and Ryan 1994).

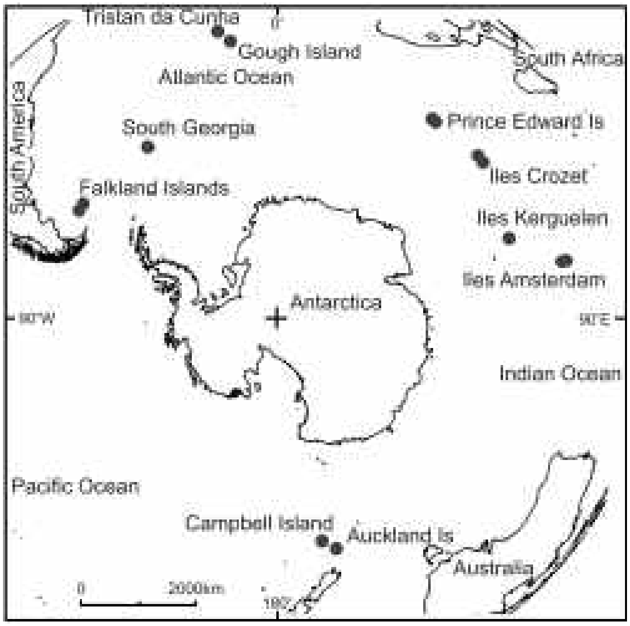

Fig. 1 Distribution of procumbent pearlwort *Sagina procumbens* on islands in the Southern Ocean.

Gough Island has never been permanently inhabited. A meteorological station on the coastal cliffs above Transvaal Bay has operated since 1963 under lease by South Africa from Tristan da Cunha. The station has a year-round staff of six to eight with an annual relief from Cape Town, South Africa, in September/October when the number of people ashore increases to 30-40 for three weeks.

Despite their remoteness, biological importance, and restrictions on access, some invasive species continue to reach these islands. In this paper, we describe the arrival on Gough Island and subsequent attempts to eradicate a localised population of the Eurasian plant, procumbent pearlwort (*Sagina procumbens*: Caryophyllaceae), a small, prostrate mat-forming herb.

SAGINA PROCUMBENS ON GOUGH

Discovery and likely source

Sagina procumbens (hereafter referred to as *Sagina*) has become invasive on at least 14 islands in the Southern Ocean, probably aided by its creeping habit, high seed production and capacity for vegetative propagation (Shaw *et al.* 2010; Fig. 1). The species was first reported from Gough Island at the meteorological station during the annual relief on 11 September 1998 (Hänel 1998). Numerous well-developed plants were then found on and around the concrete platform adjacent to the cliff crane, on concrete sections of the walkway to the main base buildings, and on the cliff near the diesel-pumping point (Hänel 1998).

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Sagina was not at that time found at localities searched farther afield along the coastline or inland.

Given the presence of a mature, post-flowering plant collected in 1998, *Sagina* had been on the island for some time before the 1997/98 summer growing season. However, annual environmental inspection reports from 1991 (when they commenced) to 1997 make no mention of any new alien plants around the station that could have been *Sagina*, despite directed checks. Plants may have been overlooked by non-botanical inspectors but the 1996 environmental inspection was undertaken by a botanist, who reported that no new introduced plants were observed around the station's buildings despite a directed search (Roux 1996). It is most unlikely that *Sagina* was present in October 1984, when Nigel Wace, a botanist with previous experience of Gough, conducted a survey of alien plants on the island (Wace 1986).

In addition to Gough Island, South Africa operates a meteorological station on sub-Antarctic Marion Island, Prince Edward Islands, in the southern Indian Ocean. Both stations are supplied through the Directorate: Antarctica and Islands, Department of Environmental Affairs (DA&I: DEA). In the 1990s, the same shipping containers were used for supplies to both islands. These containers were not always adequately cleaned before being loaded onto the supply vessel (JC pers. obs.). At Marion Island, containers unloaded from the ship by helicopter were then landed on sites inhabited by *Sagina* (Ryan 2000; JC per. obs.), which was first reported on the island in 1965 (Gremmen and Smith 1999). Some of these containers were subsequently used annually to supply Gough Island, suggesting one potential source of infestation. However, *Sagina* also occurs in the Cape Town docks area (NJMG pers. obs.), where the island cargo is loaded, so mainland Africa is also a potential source.

Sagina procumbens was first recorded on the main island of Tristan da Cunha, 380 nautical miles NNW of Gough, in 1999 on the Settlement Plain in the village of Edinburgh of the Seven Seas and within the boundary of Calshot Harbour. Subsequently, the species has been found up to 10 km from the village (Gremmen and Halbertsma 2009; JC pers. obs.). Its spread on Tristan is thought to be facilitated by human pedestrians, domestic stock and vehicular traffic. Eradication on Tristan is not considered feasible, but control was initiated in the village and Calshot Harbour in 2009 as a biosecurity measure, to reduce the risks of the plant reaching the other islands in the group (A. Rosler *in litt.* to JC 2009).

Current distribution on Gough

Since discovery in 1998, *Sagina* on Gough has spread along coastal cliffs in Transvaal Bay to its current patchy distribution over *C.* 400 m. Annual searches away from the coast, concentrating on foot paths and the less-vegetated areas in the interior, have revealed no additional plants over 10 years of effort. The very rugged nature of the island makes adequate surveys of all cliffs impossible since they reach heights of 50-300 m along most of the 42 km of coastline. However, no plants have been discovered along the island's east coast as far as 9 km from the meteorological station in Transvaal Bay. In 1999, a single *Sagina* seedling was found growing among lichens and bryophytes on a dead island tree (*Phylica arborea*) some 200 m south of the meteorological buildings. The plant may have been spread by Gough buntings (*Rowettia goughensis*) regularly seen in the vicinity (Gremmen 1999). Since then, there have been no further records *of Sagina* growing epiphytically at Gough, or outside the area described above.

The results of these surveys lead us to believe that *Sagina* is currently restricted to its known range and thus its eradication from the island should be feasible.

Prognosis of spread

On Marion Island (Prince Edward Islands), *Sagina* is spreading at a rate of 100-300 m per year (Gremmen and Smith 1999; JDS pers. obs. 2009). In May 1997, *Sagina* was found on nearby Prince Edward Island at a few sites but in subsequent years it has spread significantly (Ryan *et al*. 2003; PGR and JC pers. obs. 2008 and 2010). The plant is now considered naturalised on Prince Edward Island. Given this, and the likelihood that indigenous animals may now be facilitating its dispersal, it is considered beyond control by known methods on both Marion and Prince Edward Island.

The global distribution of *Sagina* suggests wide ecological amplitude. Although Gough is generally classified as cool temperate, its mountainous interior predominantly has herbfield and feldmark vegetation (Wace 1961) broadly similar to that of the lowlands of the sub-Antarctic Prince Edward Islands (Gremmen 1981). The precautionary view assumes that if *Sagina* becomes established in the sub-Antarctic-like uplands of Gough it will become as invasive as on the Prince Edward Islands and will then be impossible to eradicate or control.

Biosecurity procedures

Since the discovery of *Sagina* on Gough, and as part of a general improvement of biosecurity procedures within the South African National Antarctic Programme (SANAP), containers dedicated to specific islands are now used to avoid interchange between Gough and Marion. Containers are also cleaned using water under pressure inside and out before packing and loading onto the supply ship at the DA&I: DEA stores in the Cape Town docks. Once containers are loaded, the ship's holds are fumigated against invertebrates, usually the day before sailing, but this is unlikely to kill dormant seeds. The outsides of containers are also inspected visually on arrival ashore on Gough before they are opened, and their insides inspected when opened and unloaded.

ERADICATION EFFORTS ON GOUGH

On and shortly after discovery

In the year of discovery, mechanical control of *Sagina* was attempted. Plants were scraped out of cracks or removed from rock and soil surfaces, placed in strong plastic bags by volunteer members of the meteorological station and later removed from the island. Despite these measures, by the annual relief a year later (1999) the infested area had increased to about one hectare around the buildings (Gremmen 1999; Gremmen *et al*. 2001). Based on this inspection by NJMG, an eradication programme was designed and implemented in May 2000, with funding from the United Kingdom Government (Gremmen 2000).

The 2000-2004 eradication programme

A four-person team, led by NJMG, arrived on the island in May 2000 for two months (Barendse 2000; Ryan 2000). All visible plants with surrounding soil to a depth of *C.* 15 cm were removed from around the meteorological station in an attempt to remove buried seeds (Gremmen *et al.* 2001). After tests of efficacy on germination, sites where the plants had been found were treated with hot (>80°C) water from a specially designed diesel-fuelled boiler in an attempt to kill any remaining seed banks. Broad-spectrum and pre-emergent herbicides (*Glyphosate 360/Glyphogen*, *'Round-up'* and *Outpace Flowable*) were also used where it was difficult to remove plants. Lastly, hand-held blow torches were used to kill seeds in rock cracks. The use of rope-access techniques was necessary to access many of the infested sites in safety.

By the end of the 2000 visit, no plants were to be seen. Regular monitoring and herbicide spraying by volunteer team members and inspections during the annual reliefs were then viewed as the only measures required for the eventual eradication of *Sagina* from the island. A detailed manual was prepared to guide this work (Gremmen 2000). An inspection during the annual relief in September 2001 indicated that *Sagina* was under control. However, this proved to be mistaken. When JC visited the island on the 2003 relief, large coalescing clumps of *Sagina* were found at several cliff sites. The team volunteer who had been treating *Sagina* with herbicide at intervals during 2002/03 reported to JC that for safety reasons he had not ventured into all the areas where the plant was known, especially on steep and slippery cliff sections with drop-offs directly into the sea.

During the 2003 relief, another attempt was made to remove all plants for dumping at sea but there were insufficient personnel for this to be achieved. Many plants had to be left to continue growing and to flower and set seed through the 2003/04 summer, despite the efforts of the voluntary conservation officer on the meteorological team who continued to remove plants, spray herbicides and use a blow torch at intervals in infested areas throughout the year (Leveridge 2004).

Most seriously, in September 2004 wider searches for *Sagina* revealed that plants had spread northwards along the coastal cliff to a popular fishing spot known as Snoekgat, most likely through adhering to footwear (Cuthbert and Glass 2004).

Restarting and expanding the eradication programme 2005-2010

During late 2004, new funding obtained from the United Kingdom's Overseas Territories Environment Programme (OTEP) by the Tristan da Cunha Government, enabled a sustained eradication programme to be recommenced in September 2005 (Gremmen 2005; Cooper *et al*. 2006; Gremmen 2006). Because *Sagina* on Gough Island is able to set seed within three months or less from germination, it was desired to place eradication teams (with rope-access qualifications and skills) on the island for several months during each summer-growing season and at roughly quarterly intervals for long enough to remove all plants within the known distribution. In practice, such a programme was not fully achievable, primarily due to a shortage of available berths on the few vessels travelling between South Africa and the Tristan Group.

Over approximately four years, all plants found were removed and the sites treated with herbicides and/or heat during each visit. However, a few plants continued to escape detection and as a consequence flowered and set seed, thereby adding to the seed bank. This led to the prevailing situation, which since September 2008 has involved two field assistants qualified in rope access on the island for a full year. Their duties have included careful checks of the area known to be infested with *Sagina* at no more than monthly intervals, when all plants found are removed. Funding for this latest stage has again been received from OTEP, with administration of the project switching from a South African environmental consultancy (CORE Initiatives) contracted by Tristan da Cunha to the Royal Society for the Protection of Birds (RSPB), a UK-based NGO that is part of the BirdLife International partnership. The second (2009/10) team was replaced by a further two, rope access-qualified, field assistants in September 2010. Two field assistants will be appointed for 2011/12 for a fourth consecutive year. This extends the period of active eradication efforts against *Sagina* until at least October 2012. As a result of the latest protocol, very few plants have escaped notice until after they have set seed. Semi-quantitative germination trials (Visser *et al*. 2010) confirm that this has rapidly reduced the seed bank.

Expanding the 'tools in the box': new eradication techniques adopted

By 2008, despite seven years of effort, the eradication of *Sagina* on Gough had not been achieved. However, plants were being confined to a coastal distributional range, which reduced the risk of spread to the mountainous interior. Further progress required new techniques to be tested and added to those available. One new method used during the September 2008 relief was a high-pressure jet of water used to blast the peat and soil into the sea from selected infested areas, exposing bed rock. Trials in 2009 showed that salt water inhibits the germination of *Sagina* seeds (Visser *et al*. 2010). At vegetated sites, tussock grass, forbs and mosses were first removed with spades and mattocks. The vegetation and peat were then thrown or washed over the cliff edge onto the rocks below or into the sea. Subsequent checks of the newly exposed rock showed that whereas *Sagina* seedlings did continue to emerge from rock cracks they were relatively few in number, and were then easily spotted and removed. In addition, an enhanced spraying regime was commenced from October 2008 with broad-spectrum and pre-emergent herbicides applied in selected areas each month.

The soil-blasting system was not sufficiently portable for use over the full distribution of *Sagina*. In September 2009, a portable fire-fighting pump (*Davey Fire Chief*), along with a 1200-l water tank was lifted by helicopter to the northern edge of the plant's distribution at Snoekgat. A start was then made to strip the area using high-pressure hoses with a range of up to *C.* 100 m. This stripping technique is slow, labour-intensive, and may take several years to remove cover from all areas on the coastal cliffs within the range of *Sagina* down to bed rock. From September 2010, thick stands of indigenous vegetation (mainly *Spartina arundinacea* tussocks) were trimmed prior to soil blasting with a petrol-powered brush cutter. Once stripping to bed rock is completed, regular monitoring to remove seedlings soon after they germinate from rock crevices and from any small pockets of remaining soil should deplete the seed bank to zero and lead to the plant's eventual eradication from the island.

Following successful suppression of germination using salt water elsewhere (Visser *et al.* 2010), the portable pump has also been used to spray salt water (mixed in the large water tank using commercial salt brought to the island in 25 kg bags) onto the stripped rock at Snoekgat.

In addition to the new eradication attempts since September 2009, quarantine/biosecurity procedures have been strengthened in order to reduce the risks of inadvertently spreading *Sagina* inland and along the coastline. Procedures include a permanent boot wash basin at the meteorological station to ensure that footwear is cleaned of adhering soil and plant propagules plus the cleaning and inspection of containers and materials flown to food caches and camp sites in the island's interior (Gibbs 2009). These procedures are additional to the hosing down of protective clothing and footwear when leaving infested areas that has been a normal practice of the eradication campaign since its inception.

'Upping the ante': possible new techniques to test and adopt

In September 2009, an independent audit of the eradication campaign was conducted by an expert in managing alien plant eradications in South Africa (Gibbs 2009). Suggested new eradication techniques to test included salt applied in its solid form to sites where plants had been removed and the use of a helicopter-borne monsoon bucket to water-bomb the infested cliffs with salt water. The former suggestion was tested at the time, but has not proven particularly successful (Visser *et al*. 2010). The latter suggestion may be tested if an opportunity arises during annual relief visits.

Less practical suggestions included: covering the infested cliffs with a sealant material (such as the sprayed cement sometimes used to stabilise road cuttings); explosives to blast the cliff face into the sea; portable flame throwers to incinerate both plants and peat; and, probably more realistically, using some form of hormonal growth agent that would promote synchronised germination of the remaining seed bank. Weeds growing in cracks on hard surfaces can be killed with a foam surfactant created from a biodegradable glucose polymer that retains heat for longer than just water (Quarles 2001; Bridge 2005). However, hot foam would be logistically difficult to apply at any distance from the immediate surrounds of the meteorological station, given that the equipment required is not designed to be carried by hand.

The applied and proposed eradication techniques described here are not thought to place the island's indigenous biota and physical environment at any long-term risk, given that the eradication methods used are restricted to a very small part of the island

CONCLUSIONS

The eradication of *Sagina procumbens* from Gough Island has proved to be a protracted exercise. Eradication will require years of continued and concentrated effort to remove all emerging plants before they set seed, so as eventually to exhaust the existing seed bank. Biosecurity efforts to halt new propagules arriving at Gough (Lee and Chown 2009) from either Cape Town or Tristan da Cunha need to be rigorously applied, along with continued monitoring ashore to reduce the risks of the species spreading away from its current distribution. To help achieve these goals, new eradication methods and technologies should continue to be sought, tested, and adopted.

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Eradicating stoats *(Mustela erminea)* and red deer *(Cervus elaphus)* off islands in Fiordland

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Abstract In 2004, the New Zealand Government allocated NZ$7.1M to eradicate stoats (*Mustela erminea*) and red deer (*Cervus elaphus*) from Fiordland's two largest islands: Secretary Island (8100 ha) and Resolution Island (21,000 ha), in SW New Zealand. Both islands are rugged and within the swimming range of stoats and deer from the mainland. Here we apply the six strategic rules for achieving eradication to the Secretary and Resolution islands programme and use these rules as means of assessing progress five years into the campaign. For these programmes 'eradication' has been defined as the complete removal of the stoat and deer populations, and the establishment of long-term control to manage reinvasion. While the original eradication objectives are yet to be achieved, the planned conservation outcomes are on track; several threatened species of birds have been successfully reintroduced and the regeneration of palatable plants is apparent. The conservation importance of large islands such as Secretary and Resolution in terms of New Zealand's commitments to international biodiversity conventions and restoration goals cannot be overstated. However, attempting mammal eradications on such large islands in close proximity to the mainland challenges conventional paradigms for eradication. These challenges are likely to be faced increasingly by other conservation managers in New Zealand and internationally.

Keywords: Secretary Island, Resolution Island, eradication, restoration, strategic rules, extirpation.

INTRODUCTION

Fiordland National Park, in the southwest of the South Island of New Zealand, contains C. 90 islands ranging in size from small rock stacks up to Secretary Island (8140 ha), at the entrance to Doubtful Sound, and Resolution Island (20,860 ha) lying between Breaksea and Dusky Sounds (Fig. 1). The total land area of Fiordland islands exceeds 40,000 ha of which over 31,000 ha has been targeted for pest eradication.

Stoats (*Mustela erminea*) were first introduced into mainland New Zealand in the late 1880s in response to feral rabbit (*Oryctolagus cuniculus*) plagues. In 1891, Resolution Island was gazetted as one of the world's first 'reserves'. Richard Henry, curator of Resolution Island, realised the value of islands that might avoid invasion by stoats and translocated 572 birds, mostly kiwi and kakapo, to the island sanctuary. Stoats are competent swimmers (Taylor and Tilley 1984) and they had invaded many of the remote coastal islands of Fiordland only six years after their introduction to New Zealand. By 1900, Henry had confirmed the worst when he observed a stoat on Resolution Island (Hill and Hill 1987). Stoats probably invaded Secretary Island around the same time.

In 1963, the New Zealand Government designated Secretary Island a 'Special Area' within Fiordland National Park due to the island's unmodified vegetation and the complete absence of introduced browsing or grazing animals (brushtail possums; *Trichosurus vulpecula* and red deer; *Cervus elaphus scoticus*). In reality, red deer had probably already established at the northern end of Secretary Island but it was not until 1970 that a small resident population was confirmed (Mark *et al*. 1991). Control measures for red deer were implemented between 1970 and 1987 and although hundreds of deer were killed, control did not have a major impact on the population (Brown 2005). Resolution Island, also free of possums, had red deer established in high numbers by 1947 (Sutherland 1957).

Since 1999, the feasibility of eradicating island populations of stoats and managing immigration from locations within stoat swimming range has been demonstrated. Eradications of stoats from Chalky Island (514 ha) in 1999, Anchor Island (1130 ha) in 2001, and Bauza Island (480 ha) in 2002 gave managers the confidence to tackle much larger islands such as Secretary and Resolution (Elliott *et al*. 2010).

Successful eradications of pest species from islands in Fiordland have not been limited to stoats. In 2002-2007, red deer were removed from Anchor Island in Dusky Sound (Crouchley *et al*. 2011). Successful control over 50 000 ha in the Murchison Mountains (Fraser and Nugent 2003) demonstrated the feasibility of reducing the deer population to near-zero density elsewhere in Fiordland National Park and in habitats similar to those on Secretary and Resolution Islands.

The enormous potential for pest-eradication and restoration on Secretary and Resolution Islands was recognised in 2004, when the New Zealand Government allocated NZ$7.1 million over 10 years to eradicate stoats and deer from both islands. Further acknowledgement of their current intrinsic and potential future ecological values came in 2007 when they were reclassified as 'Restoration Islands' within the Fiordland National Park Management Plan (2007).

The Department of Conservation has developed an international reputation for pioneering successful single-species (rodent) eradications on remote islands (Cromarty *et al*. 2002). The next step was to expand to a 'successive culls' approach spanning many years for invasive ungulate and mustelid species. This approach was planned for Secretary and Resolution Islands and is the subject of our paper.

GOALS AND OBJECTIVES FOR FIORDLAND'S 'RESTORATION ISLANDS'

IUCN (International Union for the Conservation of Nature) guidelines define eradication as the complete removal of an alien invasive species (*IUCN Guidelines for the Prevention of Biodiversity Loss caused by Alien Invasive Species*, May 2000) whereas a programme of sustained control is focussed on managing the impacts of such species through continuous or periodic population reduction (Cromarty *et al.* 2002). In the operational

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and restoration plans for these programmes the term 'eradication' referred to the complete removal of the stoat and deer populations, and the establishment of long-term control programmes to manage reinvasion. Three goals were established: 1) eradicate stoats and deer; 2) enhance the ecological values of the islands for threatened species re-introductions; and 3) ensure that these islands remain virtually pest-free through effective island biosecurity.

Six strategic rules must be met in order for eradication to be possible (Parkes 1990; Bomford and O'Brien 1995; Parkes *et al*. 2002): 1) all target animals must be put at risk to the methods being applied; 2) target species must be killed at rates faster than their rate of increase at all densities; 3) the risk of recolonisation must be zero; 4) social and economic conditions must be conducive to meeting the critical rules; 5) where the benefits of management can be achieved without eradication, discounted future benefits should favour the one-off costs of eradication over the ongoing costs of sustained control; and 6) ideally, animals surviving the campaign should be detectable and dealt with before an increased population size becomes obvious.

The first three are regarded as crucial rules (Parkes 1990), which, unless they are met, eradication cannot proceed. Rules 4-6 are regarded as desirable (Bomford and O'Brien 1995). For example, eradication might still proceed despite social opposition. We recognised from the outset that reinvasion by stoats and deer was inevitable, so we adopted the alternative interpretation of Rule 3: the probability of the pest re-establishing is manageable to near-zero (after Broome *et al.* 2005).

Below we discuss project planning for the stoat and deer campaigns for Secretary and Resolution Islands in terms of the six rules for eradication.

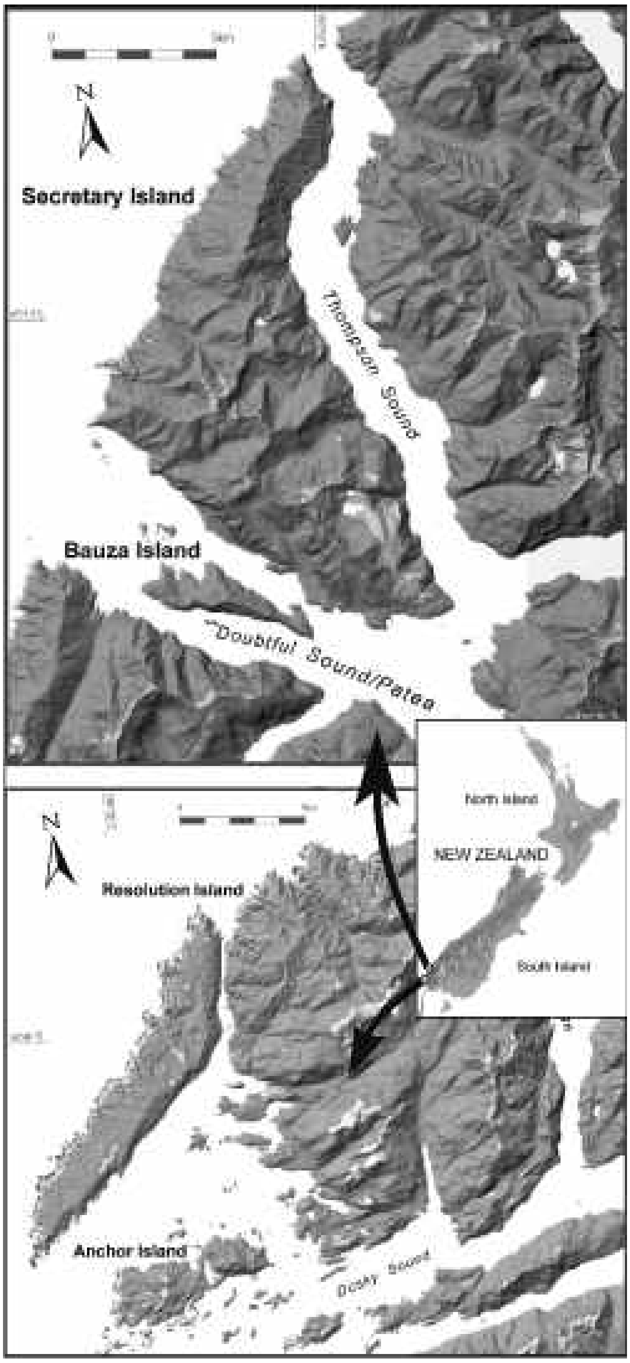

Fig. 1 Location of Secretary and Resolution Islands.

STOATS

For both islands, it seemed possible to put all stoats at risk with existing tools, tactics and strategic planning, as was detailed in operational plans by Golding *et al.* (2005) and McMurtrie *et al.* (2008). That all animals must be put at risk to the methods being applied (Rule 1), was thus considered *a priori* to hold for stoats.

Large numbers of stoats were removed in the knockdown on Secretary and Resolution Islands, but we have yet to achieve our objective of eradication (McMurtrie *et al.* 2011). A few stoats may have retained small home ranges even with the significant population reduction and have therefore never come in contact with a trap. Alternatively, a few animals may avoid entering a trap tunnel either for an extended period of time or in perpetuity (Crouchley 1994; King and Powell 2009). Rule 1, therefore, does not appear to hold for stoats on either island at the time of writing.

Stoat eradication programmes on other Fiordland islands demonstrated that animals could be killed in traps faster than their rate of increase (Rule 2), even at low densities (Elliott *et al*. 2010). With stoats, however, the real issue is not population density *per se*, but the ability to respond rapidly to 'pulsed' events such as immigration or *in-situ* breeding, particularly during mast years (Wittmer *et. al.* 2007). Rule 2 was thus considered to hold for stoats on both islands. On Secretary Island, trapping results indicate that the stoat population is being maintained at a very low level without further decline (McMurtrie *et al*. 2011) so we are not meeting Rule 2. It is too early to establish the trend for Resolution Island.

It was known from the outset that the risk of recolonisation by stoats would not be zero (Rule 3) on either island (Elliott *et al*. 2010). However, islands >300 m from a source population on the mainland were viewed as much less likely to receive immigrants than islands closer to the source population. Given that Secretary Island is 950 m from the mainland at the narrowest point, and Resolution 520 m, the risk of stoat reinvasion was assessed as low but not zero for both islands. The eradication campaign proceeded on that basis. Central to the plan was the long term use of traps used for the initial knock-down on the islands and control on the adjacent mainland, in order to manage reinvasion. Our assumptions about the rates of stoat immigration to the islands have been challenged by the results. On Secretary Island, DNA analysis of captures to June 2008 reveals a mix of residents and immigrants (McMurtie *et al*. 2011). The level of immigration detected from July 2005 to June 2008 is also higher than we predicted (see Elliott *et al*. 2010). However, unusually high immigration may have been due to a beech masting in 2006 and a subsequent rodent and stoat plague on the mainland in Fiordland. During such years, there will likely be more juvenile stoats dispersing from the mainland to inshore islands, such as Secretary Island. Further genetic work to include all of the stoats captured on both islands during the eradication campaigns should help to refine estimates for immigration.

Both the stoat and deer campaigns were initiated following a history of successful rodent eradications on increasingly large islands (Towns and Broome 2003), so public support for pest eradications in general was high (Rule 4). The deer and stoat eradication attempts on Secretary and Resolution Islands had strong iwi and community support, strong political support, and were well-funded. Rule 4 therefore held for both pest species and islands. Furthermore, ongoing support is evident within the Department and externally since much of the funding for threatened species reintroductions has come from local and national corporate sponsors. The immense conservation benefit likely to be generated by this programme has thus generated much community interest.

We also knew from the outset that Rule 5 would not hold for stoats on either island since we would never be able to disengage from the ongoing costs of sustained control. However, we calculated that near-zero density could be achieved and maintained with the same effort regardless of the number of stoats present because all of the infrastructure needed, including tracks, huts and traps, are to remain in place (and be serviced) in perpetuity. We argue that Rule 5 is not relevant where: 1) the tools and strategies for eradication are the same as those used for ongoing management; 2) there was always the intention to make continued use of kill-traps as detection and monitoring devices; and 3) the desired outcomes remain unchanged.

Another interpretation of Rule 5 is that eradication should proceed in favour of control where the benefits of the project outweigh the costs (Broome *et al.* 2005). For example, when compared with Secretary and Resolution Islands pest control to equivalent densities over 30,000 ha on mainland Fiordland would be extremely expensive without producing equivalent conservation outcomes. The existing ecological values of these islands, in particular Secretary Island which has never had introduced rodents, are unparalleled anywhere else in Fiordland in terms of scale.

Rule 6 holds for both populations of stoats because animals surviving the original knockdown campaign were largely detected and dealt with before an increased population size became obvious. Our assumption that kill-traps would provide reliable detectability was confirmed at high stoat densities using an alternative method (hair tubes) prior to the initial knock-down (Clayton *et al.* 2011). Spatial detection parameters obtained for stoats on Resolution Island using hair-tubes (Clayton *et al.* 2011) were similar to those for other published studies (Smith *et al.* 2008; Efford *et al.* 2009). However, we do not know how detectability changes with stoat density. Foot-print tracking tunnels were not used as a monitoring tool for stoats in the Secretary and Resolution campaigns because the large number required (Brown and Miller 1998; Choquenot *et al.* 2001; King *et al.* 2007) would have been prohibitively expensive and logistically difficult due to the terrain. Furthermore, any residual stoat population is likely to contain individuals that avoid tunnels, regardless of whether they contain traps or tracking cards. This observation has subsequently been confirmed by the presence of stoat tracks in snow along ridgelines with traps (McMurtrie *et al.* 2011) and video records of stoats from deer trail cameras near stoat traps on Secretary Island (D. Crouchley pers. obs.). Plans for the Secretary and Resolution Island stoat programmes did allow for the use of trained stoat-indicator dogs. We also relied on the presence of deer hunters in the four years following the stoat knock-down and their observations of stoat sign.

Because the pattern of stoat captures in kill-traps on both islands was high initially then followed by a handful of individuals in subsequent years (Clayton *et al.* 2011; McMurtrie *et al.* 2011), Rule 6 at present still holds for stoats with the caveat that information on the behaviour and detectability of stoats at low densities is imperfect.

DEER

Rule 1 was considered to hold *a priori* for deer on both islands as detailed in operational plans by Crouchley *et al.* (2007) and Crouchley and Edge (2009). On Secretary Island, an estimated 80% of the deer population was removed within the planned two-year timeframe. At the time of writing we are in the second year of the mop-up phase and therefore are yet to achieve eradication. We assume that Rule 1 still holds for deer.

We considered that the need to kill target species faster than their rate of increase at all densities (Rule 2) holds for red deer on both islands. At the time of writing, this still appears to be correct. We initially assumed that the potential for reinvasion of deer onto Secretary and Resolution islands was relatively high and that Rule 3 would not apply. This assumption has since been challenged (Crouchley *et al.* 2011) because: 1) Anchor Island has received no immigrants for the past four years despite its proximity to large deer populations on Resolution Island and the mainland; and 2) genotyping of the Secretary Island population suggested a small founder population of very few hinds and little subsequent reinvasion.

The general principle of Rule 5 was considered to hold for red deer on both islands. However, the concept of a 'one-off' campaign was rejected in favour of an ongoing programme able to be scaled down significantly once the resident population had been removed to focus on limiting reinvasion or re-establishment potential. The assumption that Rule 5 would hold for deer was planned to be addressed at a formal review in the second and fourth year of each island programme. At the time of writing it is unclear whether Rule 5 will hold for deer. Because Rules 1 and 2 hold for deer and the risk of reinvasion (Rule 3) is much lower than initially thought, eradication is likely to be achieved for deer on Secretary and Resolution Islands in the future. The alternative model is control to near zero-density akin to the Murchison Mountains (Fraser and Nugent 2003), where deer control provides massive and demonstrable conservation benefits (Burrows *et. al.* 1999; Tanentzap *et al.* 2009).

Rule 6 was considered to hold for deer on both islands. Deer are mobile and therefore leave obvious sign in many places even at low population densities (Forsyth *et al.* 2007). In addition, a variety of tools were to be employed in the mop-up phase to detect and cull deer (Crouchley *et. al*. 2011) in order to ensure complete coverage. One disadvantage of the planned deer eradication campaign was that, unlike stoats, deer control can only be implemented and/or checked regularly by people; until now, devices have not been available for continuous operation on the islands. However, there are now precedents for successful eradication of ungulates internationally using fixed devices (e.g., Ramsey *et al*. 2009).

GENERAL CONCLUSIONS

New Zealand deservedly has a reputation for successful eradication of invasive alien mammals from offshore islands. This reputation emerged primarily from rodent eradications where the risks of reinvasion were extremely low and manageable with strict biosecurity measures (Towns and Broome 2003). The Fiordland Islands programme has demonstrated that it is time to further expand our horizons to islands in close proximity (0.5 – 1 km) to the mainland, of considerably larger size than some previously attempted, and where eradication attempts involve multiple invasive alien mammal species. The conservation importance of large islands such as Secretary and Resolution in terms of New Zealand's commitments to international biodiversity conventions and restoration goals cannot be overstated. However, attempting mammal eradications on such large islands in close proximity to the mainland challenges tried-and-true paradigms for eradication. These challenges are likely to be faced increasingly by other conservation managers in New Zealand and internationally.

The eradication programmes for stoats and deer on Secretary and Resolution islands do not meet Rules 3 and 5 for eradication as defined by Parkes *et al.* (2002), but they do fit with the broader definitions as defined by Broome *et al.* (2005). The original definition of Rule 3 is applicable to offshore islands, but for islands in close proximity to the mainland, the concept of zero reinvasion risk is an ideal but not the reality with currently available tools and strategies for our focal species. Ongoing control in perpetuity becomes the only available option for stoats and deer on Secretary and Resolution Islands because of the constant, although low, risk of reinvasion. This shift in emphasis from eradication to management to zero-density is likely to become increasingly applicable to islands elsewhere as island eradication programmes worldwide tackle a range of invasive species. At this point it becomes essential to implement a strict cost-benefit analysis (Rule 5) of maintaining management to zero-density on an island in close proximity to the mainland, versus the mainland itself, where re-invasion is quicker but the site is easier to access. This requirement is especially true when funds are limited and the ongoing costs of management may be unsustainable.

Our experience to date on Resolution and Secretary Islands suggests that it is important to detect and deal with invasive animals before population sizes increase (Rule 6) for two reasons. First, the detection of individuals enables managers to mount an appropriate response, as is the case for deer. Second, if an established network of control devices (or routine hunting) doubles as ongoing surveillance and monitoring (as is the case for stoats), then animals *must* be detectable at low densities, *before* the population has increased to a level at which damage becomes a problem for threatened species and the costs of management increase. This need for a rapid response to low density populations is particularly important if there are associated threatened native species reintroduction programmes. It also highlights an important need for many threatened species in New Zealand; to quantify the relationship between population density of invasive mammals (e.g., stoats) and productivity of threatened species (e.g., fledging success; Innes *et al.* 1999) so that extra control effort can be applied should incursions result in re-establishment.

The campaigns to eradicate stoats and deer from Secretary and Resolution Islands challenge three rules for eradication, and therefore may be defined as extirpation (e.g., Parkes and Panetta 2009). Regardless of definition and the low density populations of deer and stoats, the original planned conservation outcomes have not been compromised. For example, on Secretary Island, reintroduction of mohua (*Mohoua orchrocephala*), South Island robin (*Petroica australis australis*), rock wren (*Xenicus gilviventris*), and the introduction of North Island kokako (*Callaeas cinerea wilsoni*) have proceeded as planned (Wickes and Edge 2009). Each translocation was undertaken with the knowledge that these species tolerate low density stoat populations on the mainland. Regeneration of palatable plants is already increasing in many areas as further evidence of a substantial reduction in browsing impacts by deer (Crouchley *et al.* 2011). Most planned releases of other species are still likely to proceed given the results from both island programmes to date (Wickes and Edge 2009). However, translocations of tieke or South Island saddleback (*Philesturnus C. carunculatus*) may not be possible because the species appears too sensitive to stoats at low density.

We suggest that the management of invasive mammalian species in New Zealand sits on a continuum from intensive one-off operations on offshore islands (Cromarty *et al.* 2002) through to 'local elimination' on the mainland (Morgan *et al.* 2006) (Fig. 2). The near-shore islands fall somewhere along this 'continuum of reinvasion risk.' A combination of where the programme sits on this continuum and how it fulfils the conservation objectives under Rule 5 is the main consideration when attempting a programme of this nature.

The following lessons arose from the stoat and deer eradication programmes in Fiordland:

Smaller to larger scales. There are international precedents for learning from eradication of top predators and ungulates on islands (e.g., Ramsey *et al.* 2009). Before we made attempts at a larger scale, developmental information vital to the success of the Secretary and Resolution Islands programme came from smaller Fiordland islands, including the likelihood of immigration from the mainland (Elliott *et al.* 2010; Crouchley *et al.* 2011).

Shorter to longer time frames. Eradication of stoats from some of the smaller Fiordland islands was carried

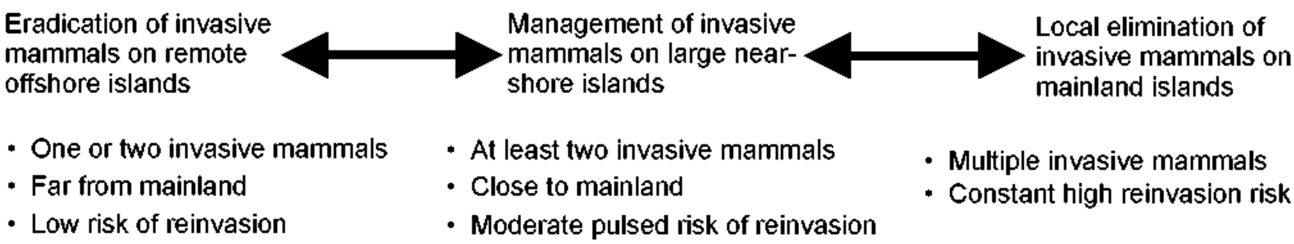

Fig. 2 Continuum of reinvasion risk from remote offshore islands to local elimination on the mainland.

out on a relatively short time scale (weeks to months). A key distinction in the attempted eradications of stoats from Secretary and Resolution is the planned extended time frame both for pest removal and subsequent reintroductions of threatened native species (10 years). This extended time scale links with the increased spatial scale (above) and is a commonsense approach to invasive species management on larger islands.

Control tools double as surveillance devices. The tools used in initial knock-down of stoats and deer on Fiordland islands, which involved kill-traps for stoats, and helicopter and ground hunting for deer, are the same tools used for ongoing monitoring and surveillance of both species. This approach is an efficient and cost-effective use of limited funds that allows conservation managers the strategic option of applying the same tools during the maintenance phase of the programme.

Early adoption of new technology. As the focus for an eradication campaign shifts from population knock-down to targeting individuals, the deer programme in particular has shown the value of an iterative process in developing and applying technology such as trail cameras, remote monitored deer pens and telemetered animals (Crouchley *et al.*2011). These tools significantly improve success through increased understanding of behaviours for animals at low population density and specific to the site.

Flexibility in planning in order to respond to new knowledge. Extirpation to zero-density of ungulates and mustelids through successive culls is not the same as a one-off rodent eradication. Not surprisingly, the biology of the species involved has played a key role in defining success, and our initial assumptions as to how each species would respond to an eradication attempt were sometimes incorrect. To that end, 'successive cull' jobs require flexible plans that adapt to events and results as the project proceeds (Parkes and Panetta 2009). For deer, early attempts at eradication shaped thinking for over two decades as to what might be achievable. For stoats, it was initially assumed that even one or two stoats remaining on (or arriving on) these islands was unacceptable. For both these species, our thinking changed as we learned more about both the history of eradication attempts, rates of reinvasion, and the achievability of eradication, given currently available tools. Furthermore, for both species there has been a shift from the need to quantify abundance (number of animals present) to quantify detection probabilities at low densities. Understanding how fundamental ecological parameters such as home range size, movements, genetic relatedness, and detectability change throughout the campaign, and information on how these parameters vary through time or among individuals, can considerably enhance knowledge for future eradication efforts, and has profound implications for how the operations are planned.

The need for clearly-stated objectives and a continued focus on the restoration goals. Eradication of invasive mammal species from islands in Fiordland involved clear statements of objectives in the operational and restoration plans. In addition, programme objectives have been reviewed at key times, and restoration work aligned with project milestones. Even if extirpation or 'management to zero-density' does not sound as compelling to the community as 'eradication', the outcomes through reintroduction of threatened or endangered species, population responses of *in situ* native biota, and ecosystem responses are extremely compelling and desirable.

ACKNOWLEDGEMENTS

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* Unpublished reports and operational plans available at *www.doc.govt.nz/conservation/land-and-freshwater/australasias-top-25-restoration-projects/*

Targeting multiple species – a more efficient approach to pest eradication

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Abstract To date, most eradications of introduced mammals on islands have targeted a single species or a subset of the pest species present. More recently, this approach has changed to simultaneously target suites of pest species. To assess the relative efficiency of the two approaches, I reviewed and compared successful multi-species and single species eradications of stoats (*Mustela erminea*), cats (*Felis catus*), brushtail possums (*Trichosurus vulpecula*) and rabbits (*Oryctolagus cuniculus*). Multi-species pest eradications that targeted cats, stoats and possums often achieved a successful result in a shorter period of time and required less effort than single species operations targeting the same species. In contrast, I found no difference for operations that targeted rabbits. I consider primary and secondary poisoning to be the major factors contributing to the increased operational efficiency observed, but other influences may also have played a part. The results from the 39 eradication operations reviewed suggest that for some species the multi-species approach, although more complex, is more efficient and may ultimately reduce the cost of an eradication programme.

Keywords: Island, invasive species, cats, *Felis catus,* stoats, *Mustela erminea,* brushtail possums, *Trichosurus vulpecula,* rabbits, *Oryctolagus cuniculus,* secondary poisoning

INTRODUCTION

The first eradications of invasive mammals on islands in New Zealand were undertaken in the early 1900s (Clout and Russell 2006). These operations targeted individual species, an approach that would endure for the next eighty years. The successful eradication of Norway rats (*Rattus norvegicus*) and house mice (*Mus musculus*) on Whenuakura Island in 1983 (Newman 1985), and the removal of Norway rats in 1985 from Moutohora Island during an eradication campaign against rabbits (*Oryctolagus cuniculus*) (Towns and Broome 2003), indicated the potential for simultaneous eradications of multiple pest species.

By monitoring rodent populations during poisoning operations against brushtail possums (*Trichosurus vulpecula*) on the New Zealand mainland, Innes *et al.* (1995) demonstrated the effects of poisoning operations on non-target pest species and provided the scientific basis for targeting several pest species with the same technique. The first attempts to eradicate multiple species on islands were carried out on a small scale. Rabbits and Pacific rats (*R. exulans*) were removed in the same poisoning operation on Korapuki Island (18 ha) in 1986 (McFadden and Towns 1991) and possums and mice were eradicated from Allports Island (16 ha) in 1989 (Brown 1993).

The potential for exploiting secondary poisoning as a means of controlling invasive predators in New Zealand was revealed in the 1990s when the fate of stoats (*Mustela erminea*) and cats (*Felis catus*) was monitored during campaigns using poison to control rodents, rabbits or possums (Alterio *et al.* 1997; Murphy *et al.* 1999). The first island operation to take advantage of these secondary effects, on Tuhua (Mayor Island) (1277 ha) in 2000, successfully eliminated Norway rats, Pacific rats and cats (Williams and Jones 2003).

More recently, eradication attempts have aimed at increasingly large areas and a greater diversity of species (Murphy *et al.* 1999; Speedy *et al.* 2007). Eradications within pest-proof fenced areas on the New Zealand mainland have in some cases removed more than 10 pest species in the same operation (Speedy *et al.* 2007). The most recent New Zealand multi-species eradication operation, which is currently underway on Rangitoto and Motutapu islands in the Hauraki Gulf, aims to simultaneously remove from 3854 ha seven mammalian pests: ship rats (*Rattus rattus*), Norway rats, mice, stoats, rabbits, cats and hedgehogs (*Erinaceus europaeus occidentalis*).

In this paper I test the hypothesis that multiple species eradication operations are more efficient than those targeting a single species. I present information and results from 39 eradications to help illustrate factors that may contribute to greater operational efficiency. Reasons, in addition to primary and secondary poisoning, are given as to why the multiple species operations reviewed may have been more efficient.

METHODS

For this analysis, only operations against cats, stoats, possums and rabbits were reviewed because insufficient data were available for other species. Operations were divided into those that targeted more than one species (multi-species eradications) and those aimed at a single species. Cats and stoats were included as examples of invasive predators, whereas possums and rabbits were chosen because both are primarily herbivores (King 2005).

I used the length of time taken and the number of trap nights per hectare completed to provide an indication of operational efficiency. The time taken to complete eradications was measured in months and was defined as the time from when the operation began to when the last animal or sign of the target species was seen. It did not include the time spent pre-baiting or monitoring to confirm eradication success.

Trap nights were used as a measure of effort, but only for cats, stoats and possums. Insufficient data precluded a comparison of the cost of eradication and the hunting effort required against rabbits. Trap nights were counted from when traps were first set to when the last animal was trapped or the last sign of an animal was recorded. While terrain differences could not be accounted for, the great variation in size between sites was corrected by presenting trap nights as trap nights per hectare. There was some variability in trapping protocol, but I consider trap nights per hectare to be a relatively reliable measure of effort for comparison between the operations reviewed.

Comparisons were restricted to operations that used a similar range of techniques effectively excluding operations undertaken before 1970. To further counter the influences of island size and population viability, operations that targeted cats at sites less than 1000 ha were excluded and a minimum area of 100 ha was prescribed for sites where stoats, possums and rabbits were targeted. Because the nature of the data was non-parametric, the Mann-Whitney *U* test was used to compare between groups. Samples were independent, so met the conditions necessary for this test.

RESULTS

Nine operations against cats were reviewed, five of which were part of multi-species projects and four that specifically targeted cats (Table 1). All nine employed poisoning and live trapping. Both the Marion Island and the Little Barrier Island operations were carried out in the 1970s, but techniques for trapping and poisoning cats have not changed significantly since (van Aarde 1980; Veitch 2001; Ambrose 2006). All of the multi-species operations eliminated cats over a shorter timeframe (Fig. 1) and required significantly fewer trap nights per hectare (Fig. 2) than the cat specific operations.

Eight successful operations against stoats were carried out at sites greater than 100 ha (Table 1), all of them within New Zealand. Two were mainland eradication projects completed within areas protected by a pest proof fence. Kill trapping was undertaken in all eight projects and in all three multi-species operations, rodents were targeted first with the aerial application of rodent bait. All of the stoat specific operations were undertaken on islands with no other introduced pest species present and were completed within 12 months. However, all three multi-species operations eliminated stoats as a consequence of rodent eradication requiring, on average, less time and significantly less trapping effort (Figs 1 and 2).

Only five operations against possums at sites >100 ha in area were available for review (Table 1). Poisoning and live trapping were the principal methods used. Although not statistically significant, the three multi-species operations eliminated possums more rapidly than where possums were the only target (see Fig. 1). The number of trap nights was also markedly reduced (Fig. 2).

Seventeen successful rabbit eradications on islands over 100 ha were reviewed (Table 1). Sites were more geographically widespread than for the other species and whilst most operations utilised poisons, several relied solely on hunting and trapping. No discernible difference in time was apparent between these single species and multi species operations (see Fig. 1). Other indications of effort such as hours spent hunting or trapping were unavailable for sufficient operations to permit comparison.

For some of the multi-species eradications, using techniques to target more than one species, appears to have increased operational efficiency. Tracking tunnels used to detect rodents at Maungatautari were instrumental in determining the location of surviving cats (Speedy *et al.* 2007) (Table 1). Cats were opportunistically targeted while spotlighting for rabbits during the Rangitoto and Motutapu pest eradication and trapping for cats caught one of the last two surviving rabbits hastening the elimination of these species. The eradication of possums and wallabies (*Petrogale penicillata penicillata*) from Rangitoto and Motutapu in the 1990s also beneficially exploited the susceptibility of two target species to the same techniques (Mowbray 2002).

Multi-species operations against similar suites of species with the same techniques did not always produce the same outcome. Although possums were eliminated by an aerial application of bait to eradicate rodents over 3300 ha at Maungatautari (Speedy *et al.* 2007), two possums survived a similar operation over 252 ha at Karori Sanctuary (R. Empson pers. comm.). Cats were successfully eliminated as a consequence of the rodent eradication on Tuhua but not on Raoul Island (Ambrose 2006) with the same suite of species. Hedgehogs disappeared at Maungatautari following the application of rodent bait (Speedy *et al.* 2007) but persisted on Rangitoto and Motutapu (Griffiths 2010).

DISCUSSION

The hypothesis that targeting multiple species is a more efficient approach to cat and stoat eradication was supported by: 1) significantly less time and trapping effort required to eradicate cats when this species was targeted amidst a suite of pests, and 2) the successful elimination of stoat populations as a by-product of rodent eradications. Whether the same can be applied to other invasive predators remains to be seen. However, evidence that invasive

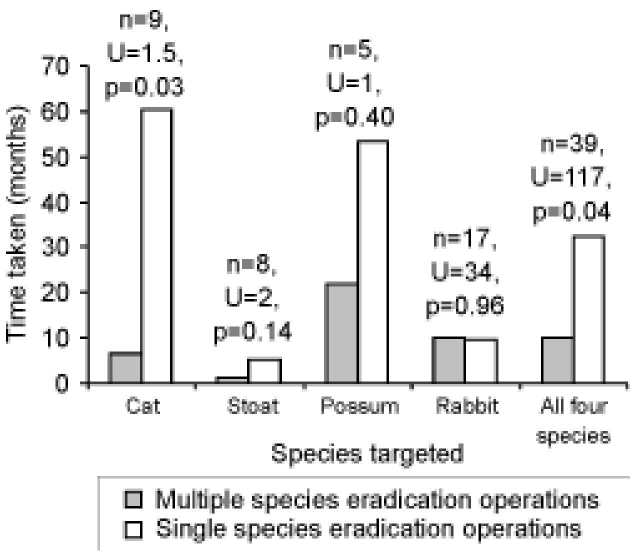

Fig. 1 Comparison of the time taken by multi-species and singles species eradication operations to successfully eliminate cats (*Felis catus*), stoats (*Mustela erminea*), brushtail possums (*Trichosurus vulpecula*) and rabbits (*Oryctolagus cuniculus*). *Statistical values were derived from the Mann-Whitney* U *test and are displayed to two decimal places.*

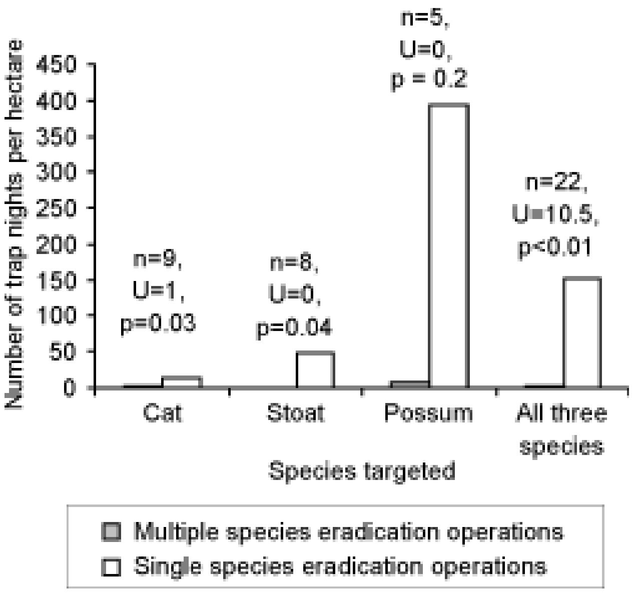

Fig. 2 Comparison of the trapping effort required by multi-species and singles species eradication operations to successfully eliminate cats (*Felis catus*), stoats (*Mustela erminea*) and brushtail possums (*Trichosurus vulpecula*). *Statistical values were derived from the Mann-Whitney* U *test and are displayed to two decimal places.*

Table 1 Multi-species and single species eradications that have successfully eradicated cats (*Felis catus*), stoats (*Mustela erminea*), brushtail possums (*Trichosurus vulpecula*) and rabbits (*Oryctolagus cuniculus*). *The author would be grateful to be made aware of any omissions or errors in this compilation. Methods listed are: B=biological control; P=poison; T=trapping; H=hunting; D=dogs.*

Target spp	No spp	Site	Area (ha)	Operation start date	Last animal or sign recorded	Months to complete *	Trap nights/ha required	Methods used	Refs
Cat	1	Macquarie Island[1]	13,182	May 1998‡	Jun 2000	14	14.382	P, T, H	16
Cat	1	Little Barrier (Hauturu)	3083	Jul 1977	Jun 1980	36	25.638	P, T, H	27
Cat	1	Marion Island	29,000	Mar 1977	Jan 1991	168	12.402	B, P, T, H, D	5
Cat	1	Ascension Island	9700	Feb 02	Jan 2004	23	4.36	P, T, H	23
Cat	7	Rangitoto & Motutapu	3850	Jun 2009	Sep 2009	4	0.182	P, T, H	14
Cat	3	Raoul Island	2938	Jul 2002	Jun 2004	23	5.667	P, T, D	2; 3.
Cat	3	Mayor Island (Tuhua)	1277	Sep2000	Oct 2000	2	0	P	28
Cat	2	Hermite Island[2]	1020	Jul 1999	Aug 1999	2	1.514	P, T	1
Cat	15	Maungatautari	3300	Sep 2004	Oct 2004	2	0.004	P, T	8.
Stoat	1	Anchor Island	1280	Jul 2001	Nov 2001	4	54.423	T	13
Stoat	1	Te Kakahu (Chalky) I.	511	Jun 1999	Oct1999	5	181.259	T	30
Stoat	1	Bauza Island	475	Jun 2002	Jun 2003	12	299.3	T	29
Stoat	1	South Passage Island	176	Jun 1999	Oct1999	5	1.948	T, D	30
Stoat	1	Doubtful Islands	120	Jan 2000	Feb 2002	1	4.631	T	13
Stoat	15	Maungatautari	3300	Sep 2004	Sep 2004	1	0	P	24
Stoat	7	Rangitoto & Motutapu	3850	Jun 2009	Aug2009	2	0	P, T	14
Stoat	10	Karori Sanctuary	252	Sep 1999	Oct 1999	1	0	P	22.
Possum	1	Kapiti Island	1965	Feb 1980	Oct 1986	69	531.456	P, T, D, S	7
Possum	1	Codfish (Whenua Hou)	1396	Feb 1984	Apr 1987	38	256.59	P, T, D	6
Possum	15	Maungatautari	3300	Sep 2004	Oct 2004	2	0	P, T	24
Possum	10	Karori Sanctuary	252	Sep 1999	Nov 1999	2	0.04	P, T	22.
Possum	2	Rangitoto & Motutapu	3850	Nov 1990	Dec 1996	61	25.638	P, T, H, D	19
Rabbit	1	Bird Island	101	Oct 1996	Nov 1996	1	N/A	P	18
Rabbit	1	Broughton Island	144	May 1997	Sep 1997	5	N/A	B, P	11.
Rabbit	1	Enderby Island	710	Feb 1993	Apr 1993	2	N/A	P, T, H, D	25
Rabbit	1	Ile aux Cochons	165	Jul 1997	Jun 2000	35	N/A	P, T, H	10
Rabbit	1	Ile Guillou	145	Jul 1994	Dec 1995	18	N/A	P, H	10
Rabbit	1	Île Verte	148	Jul 1992	Jan 1994	18	N/A	P, H	10
Rabbit	1	Isla Deserta Grande	1206	Sep 1996	Dec 1996	3	N/A	P	4
Rabbit	1	Lehua Island	120	Nov 2005	Feb 2006	3	N/A	T, H, D	9
Rabbit	1	Round Island	196	Jul 1986	Sep 1986	2.5	N/A	P, H	17
Rabbit	1	San Benitos West	547	Jan 1998	Sep 1998	9	N/A	T, H	12
Rabbit	2	Motuihe	160	Jun 2002	Apr 2004	22	N/A	P, T, H, D	21.
Rabbit	2	Moutohora (Whale)	143	Aug 1985	Sep 1987	25	N/A	P, T	15
Rabbit	2	Salvagen Grande	240	Aug 2002	Aug 2002	1	N/A	P	31
Rabbit	2	Stanley Island	100	Sep 1991	Sep 1991	1	N/A	P	26
Rabbit	2	Todos Santos Sur	100	Nov 1997	Jul 1998	8	N/A	T, H	12
Rabbit	3	St Paul Island[3]	800	Feb 1997	Feb 1999	3	N/A	P, H, D	20.
Rabbit	7	Rangitoto & Motutapu	3850	Jun 2009	Mar 2010	10	N/A	P, T, H, D	14

[1] Extensive cat control had taken place on Macquarie up to this point.
[2] The Hermite operation targeted cats after an unsuccessful rat eradication project. However, rodent numbers were greatly reduced and both primary and secondary poisoning were still a probable consequence.
[3] Although the St Paul operation spanned two years, the eradication operation was completed in three short field trips.

References: 1. Algar *et al.* 2002; 2. Ambrose 2006 ; 3. Ambrose *pers. comm.*; 4. Bell 2001; 5. Berthier *et al.* 2000; 6. Brown 2002; 7. Brown and Sherley 2002; 8. C. Speedy *pers. comm.*; 9. Campbell *pers. comm.*.; 10. Chapuis *et al.* 2001; 11. D Priddel *pers. comm.*; 12. Donlan *et al.* 2000; 13. Elliot *et al.* 2010; 14. Griffiths 2010; 15. Imber *et al.* 2000; 16. K. Springer *pers. comm.*; 17. Merton 1987; 18. Merton *et al.* 2002; 19. Mowbray 2002; 20. N Torr *pers. comm.*; 21. P Keeling *pers. comm.*; 22. R Empson *pers. comm*.; 23. Ratcliffe *et al.* 2009; 24. Speedy *et al.* 2007; 25. Torr 2002; 26. Towns and Broome 2003; 27. Veitch 2001; 28. Williams and Jones 2003; 29. Willans 2003a; 30. Willans 2003b; 31. Zino *et al.* 2008 predators such as mongoose (*Herpestes javanicus*) and foxes (*Vulpes vulpes*) are also susceptible to secondary poisoning (Braverman 1979; Berny *et al.* 1997) suggests that similar efficiencies could be gained.

All of the multi-species operations that involved cats and stoats began with the application of rodent bait containing brodifacoum targeting rodents (Table 1). Secondary poisoning, as described by Alterio *et al.* (1997), is the most likely mechanism to have eliminated stoats and reduced cat populations resulting in the reduced investment in time and trapping effort but other factors such as primary poisoning may also have assisted. In the Tuhua operation the first cats were found dead seven days after the application of rodent bait (Williams and Jones 2003) suggesting that factors other than secondary poisoning were involved.

Individual predators that do not succumb to secondary poisoning should become more susceptible to follow up techniques such as trapping when an important prey item is removed. However, this was not apparent from the operations reviewed. The cats trapped or shot on Raoul Island and Rangitoto and Motutapu following the eradication of rodents were all in good condition with substantial fat stores indicating these individuals had access to alternative sources of prey (Ambrose 2006; *pers. obs.*). On Rangitoto and Motutapu, cats scavenging carcasses of dead non-target wildlife were apparently not eating the internal organs where anticoagulant residues are highest or stomachs containing undigested bait (Dowding *et al*. 1999). It is also possible that the diet and behaviour of surviving individuals was in some way different from the rest of the population. Further research on why some individuals in a population survive when others do not is required.

Reduced time and trapping effort was apparent for the multi-species operations that also targeted possums. The aerial application of cereal bait containing sodium monofluoroacetate (1080) is commonly used against possum populations on the New Zealand mainland (Cowan 2005) and the eradication of possums at Maungatautari and near elimination of possums at Karori as a consequence of rodent eradication can be directly attributed to primary poisoning. Many other non-target herbivorous or semi-herbivorous pest species such as deer (*Cervus spp*.) and pigs (*Sus scrofa*) have also been shown to be vulnerable to primary poisoning (Innes and Barker 1999) and other invasive species are likely to be similarly affected.

Rabbits are susceptible to the cereal baits used to eradicate rodents, and rabbits have been eliminated through poisoning alone (Bell 2001; Zino *et al*. 2008; Towns and Broome 2003). However, no time savings were apparent for the multi-species operations that included rabbits. In most of the operations reviewed, some individuals survived the initial poisoning campaign (e.g., Torr 2002; Micol and Jouventin 2002), necessitating detection and elimination of survivors. How quickly this can be achieved for rabbits may be independent of whether other species are targeted.

The use of the same technique against more than one pest species also appears to have contributed to the increased efficiency of some multi-species eradications. At sites where introduced prey cannot be targeted first, the use of complementary techniques may be the most significant factor influencing efficiency. However, there are risks to using the same methods against multiple species. For example, efficiency gains could be undermined by a failure to prioritise between target species or resources diverted by the need to deal with wary survivors.

In conclusion, comparisons between multi-species and single species eradication operations support the hypothesis that the multi-species approach can be more efficient for some species. Multi-species operations that have targeted cats, stoats and possums have not only simultaneously eradicated several species, they have also achieved eradication in less time and with less effort than those against single species. Primary and secondary poisoning of non-target pest species are likely to have been the most important factors reducing the time and effort required, but other factors such as the use of complementary techniques may also have contributed. The cost of the eradication operations reviewed in this paper was unable to be determined, but the duration of operation and trapping effort required are likely to be indicators of resource investment. By targeting a suite of species in the same operation it is expected that, on a species by species basis, resources will be conserved.

Eliminating an invasive predator in the same operation as removing prey is also likely to minimise the risk of prey switching (Innes and Barker 1999), a particularly valuable consequence if vulnerable native species are present. The removal of invasive predators also removes the possibility of meso-predator release effects such as those described by Rayner *et al.* (2007).

With the benefit of hindsight, operations such as the eradication of cats on Macquarie, Little Barrier and Marion islands, and possums from Kapiti and Codfish islands, may have been achieved more quickly with less trapping effort required and at reduced cost if other resident pest species such as rodents or rabbits had been targeted first. However, when these operations were undertaken, the technology for eradicating rodents and rabbits across such large areas had not yet been developed (Towns and Broome 2003) and the pressure on native species from these pests was at the time considered to be unsustainable (Veitch 2001;van Aarde *et al.*1980). The conclusions of this report are of more relevance to future projects such as the proposed eradication of rodents and cats on Great Barrier Island (Ogden *et al.* 2011) and the planned removal of stoats from Resolution Island (McMurtrie *et al.* 2011).

The persistence of cats on Raoul Island and hedgehogs on Rangitoto and Motutapu provides a precautionary end to end this paper. Both of these species have been eliminated as a consequence of rodent eradication elsewhere (Ambrose 2006; Speedy *et al.* 2007). While it is possible to make inferences based on the outcomes of other pest eradications, it is always possible that the results obtained in one location will not translate to the same outcomes elsewhere. Consequently, eradication project managers must always plan conservatively and, for many species, anticipate the survival of individuals. For this reason, intensive monitoring for survivors will continue to be a critical component of any eradication programme.

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Eradicating multiple pests: an overview

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Abstract Of 432 marine islands worldwide where invasive vertebrate eradications have taken place, 332 (77%) were of a single species. Up to five species have been eradicated on a further 99 islands, and eight were removed from Kapiti Island, New Zealand (although these eradications spanned 82 years, from 1916 to 1998). In contrast, some recent eradications from fenced 'mainland islands' in New Zealand have removed 10-12 species in a few months, enabling significant new restoration opportunities. We suggest that multi-species eradications arose a) because there is growing confidence to eradicate more species (not just vertebrates) from more environments; b) to enable broader ecological restoration (at community/ecosystem level); c) to overcome unhelpful ecological responses of remaining invasive plant or animal species to the removal of a single predator or competitor, and d) in New Zealand in particular, because there is a rapidly growing network of community-driven restoration projects on and near the inhabited mainland, with a larger pest mammal suite. Multi-species eradications, particularly in mainland situations, tend to have many stakeholders, broad ecological goals, complex eradication and monitoring requirements, but high benefit:cost ratios per eradicated species. Remaining challenges are to clarify key ecological, social and cultural goals; to prioritise sites; to determine the order in which different taxa should be targeted; to work with increasing numbers of stakeholders as eradications increasingly occur in inhabited places; to increase their physical scale, and to build capacity and knowledge.

Keywords: Eradication, vertebrate pests, islands, fenced sanctuaries, ecological restoration

INTRODUCTION

Invasive plants and animals – particularly mammals – are a major cause of ecological disruption, species extinction and economic loss throughout the world, especially on islands (Atkinson 1989; Vitousek *et al.* 1997; Chapin *et al.* 2000; Mack *et al.* 2000). However the number of successful eradications substantially increased in the 1980s and 1990s, particularly on islands where low reinvasion risk and few stakeholders together increased chances of success (Towns *et al.* 1997; Myers *et al.* 2000; Simberloff 2002; Clout and Russell 2006).

These successes are exemplified by eradications of rats from increasingly large and diverse islands, culminating in the removal of Norway rats (*Rattus norvegicus*) from 11,330 ha Campbell Island in 2001 (Clout and Veitch 2002; Clout and Russell 2006). Eradications have also targeted an increasing diversity of species. The Global Island Invasive Vertebrate Eradication Database documents 949 attempts against 37 species of introduced vertebrates (Keitt *et al.* 2011), with 432 identified as successful (http://db.islandconservation.org/ accessed 28 January 2010). Of the successes, 332 were of one species; 65 were of two species; 22 were of three species; eight were of four species; four were of five species and one, on Kapiti Island, New Zealand was of eight species. The latter eradications included cattle (*Bos taurus*) in 1916, goats (*Capra hircus*) in 1928, sheep (*Ovis aries*) in 1930, cats (*Felis catus*) in 1934, brushtail possums (*Trichosurus vulpecula*) in 1986, and Norway and Pacific rats (*Rattus exulans*) in 1996. The eradications on this island thus spanned 82 years.

In contrast, since 1999 up to 14 introduced mammal species have been eradicated over a few months from fenced 'mainland' sanctuaries in New Zealand at Karori (225 ha), Maungatautari (3400 ha), Rotokare (230 ha) and Orokonui (315 ha). In addition to those eradicated from Kapiti, the species removed were ship rats (*Rattus rattus*), stoats (*Mustela erminea*), ferrets (*M. furo*), weasels (*M. nivalis*), hedgehogs (*Erinaceus europaeus*), rabbits (*Oryctolagus cuniculus*), brown hares (*Lepus europaeus*), feral pigs (*Sus scrofa*), red deer (*Cervus elaphus*) and fallow deer (*Dama dama*). The significance of these sites is that they targeted a large suite of invasive mammals, included species with global distribution, and were rapidly achieved.

In this paper we briefly examine the origins of such multi-species eradications, primarily using New Zealand examples. We discuss how they differ from eradications of single species, and consider the key challenges facing the eradication of several species at the same time.

THE ORIGINS OF MULTI-SPECIES ERADICATIONS

Growing confidence to eradicate more species in more environments

Every successful eradication increases confidence for further projects, especially when they have been the first to eradicate invasive plants or animals or have been undertaken in new environments. Successful eradications have now been achieved for 28 species of mammals and nine species of birds (http://db.islandconservation.org/ accessed 26 March 2010) as well as some amphibians and reptiles (Rodda *et al.* 2002; Beachy *et al.* 2010; Orchard 2010). Other efforts have also targeted insects (Allwood *et al.* 2002; Krushelnycky *et al.* 2002) and plants (e.g., Coulston 2002; West 2002).

Many eradication advances have been orchestrated rather than accidental. Towns and Broome (2003) list the advances made by strategic eradication attempts against rats on New Zealand islands from 1988 to 2001. The 2001 eradication of Norway rats from Campbell Island, New Zealand, was at that stage 'the largest, most isolated and most logistically challenging rodent eradication attempted' (McClelland 2010). Such success encourages rodent eradications on other large or remote islands. Experiments with bait interference by land crabs, and extensive canopy use by rats, are being undertaken to allow the successful application of eradication techniques learned in temperate and subantarctic regions to be applied to the tropics (Wegmann *et al.* 2010).

Unhelpful interactions between pest species

Eradicating or suppressing invasive mammals in complex and altered ecosystems risks unexpected ecological outcomes, through compensatory changes in the abundance, behaviour and thus impact of remaining exotic populations (Towns *et al.* 1997; Zavaleta *et al.* 2001; Zavaleta 2002). The pathways for such interactions are now fairly well understood through multi-species control in New Zealand mainland sanctuaries. A typical suite of small mammals at such sites includes carnivores (feral cats, stoats) and omnivores (brushtail possums, ship rats and mice; *Mus musculus*). Controlling cats or stoats alone is likely to increase ship rats by mesopredator release (Efford *et al.* 2006; I. Flux and C. Gillies, unpub. data). The increase of mice after effective ship rat control (Innes *et al.*1995) is also probably due to release from predation. Ship rats also proliferate when brushtail possums are controlled, presumably by release from exploitation

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competition for the fruits, seeds and invertebrate foods (Sweetapple and Nugent 2007). Predators may also respond to loss of a preferred prey by switching diet, sometimes to valued native prey. Stoats in North Island (New Zealand) forests mainly eat ship rats, and when rat densities are reduced, they may eat more birds (Murphy and Bradfield 1992). Similar interactions have been found on New Zealand islands. Regardless of site, the order in which species are eradicated should be considered before an eradication commences (Morrison 2010). Alternatively, the simultaneous eradication of interacting species can overcome these problems (Roy *et al.* 2002).

Interactions between plants and herbivores are also common. Mammals that damage native plants can also keep weeds in check. For example, exclosures established on Rangitoto Island, N.Z., before brushtail possums and wallabies (*Petrogale pencillata pencillata*) were eradicated, showed that pampas grass (*Cortaderia selloana*) and prickly hakea (*Hakea sericea*) may increase when possums and wallabies were removed (Wotherspoon and Wotherspoon 2002). On Raoul Island, fruiting by native and exotic species increased after rat removal. This led West and Havell (2010) to subsequently recommend changes to the management of the exotic plants *Catharanthus roseus* and *Bryophyllum pinnatum* from surveillance to eradication. Following the eradication of rabbits from Lehua Island, Hawaiian Islands, some native plant species increased in abundance but native plant cover decreased overall, due mainly to the spectacular spread of introduced shrubs and the grasses *Setaria parviflora* and *Cenchrus ciliaris* (Eijzenga 2010).

Broadening goals of ecological restoration

Eradications are not ends in themselves, but a strategy for achieving ecological restoration. There has been increasing recognition in New Zealand during the last three decades of the broader recovery of whole *ecosystems*, rather than simply the recovery of threatened species *populations* (Towns *et al.* 1997). Broader restoration goals in turn demand broader restoration actions, including multi-species eradications.

Lee *et al.* (2005) reviewed national and international frameworks for biodiversity monitoring and suggested that the 'primary national outcome of conservation management at the highest level is to maintain ecological integrity, here defined as the full potential of indigenous biotic and abiotic features, and natural processes, functioning in sustainable communities, habitats, and landscapes'. Key elements of ecological integrity are:

Indigenous dominance – the level of indigenous influence on the composition, structure, biomass, trophic and competitive interactions, mutualisms, and nutrient cycling in a community;

Species occupancy – the extent to which any species capable of living in a particular ecosystem is actually present at a relevant spatial scale, and

Environmental representation – the distribution of indigenous biota across environmental gradients derived from data layers based on climate, soils and geology (Lee *et al.* 2005).

In this powerful framework, the ecological goal of eradication of exotic species is to restore indigenous dominance, for example so that indigenous biomass 'diverted' to exotic consumers is restored to original indigenous consumers. Translocations to predator-free islands can restore species occupancy by placing taxa back at a place where they formerly existed, thus restoring the ecological processes and mutualisms in which the taxa were formerly participants. Towns *et al.* (1997) distinguish between restoration of mainland communities on islands (*ex situ* restoration of threatened mainland taxa) and restoration of island communities *per se* (*in situ* restoration).

The restoration of these processes can have complex and counter intuitive results. On Kapiti Island (New Zealand), invertebrate catch frequency unexpectedly declined three years after Norway and Pacific rats were eradicated, perhaps because native birds released from rat predation and competition were more effective invertebrate harvesters than the rats (Sinclair *et al.* 2005). The key change is that the resultant predation processes, pressures and outcomes have now become dominated by native species and locally derived ecological relationships.

Perhaps the most sophisticated exploration of ecosystem impacts of island invaders was recently documented for 18 islands off northern New Zealand. Here, invasion by ship and Norway rats significantly reduced seabird abundance by predation, which consequently increased plant litter depth, and reduced forest soil fertility by disrupting the transport of nutrients from sea to land by seabirds. This in turn generally reduced the abundance of below-ground organisms and changed the ecological processes they mediated, while above-ground plant biomass was greater when rats were present (Fukami *et al.* 2006; Towns *et al.* 2009). Rat invasion also reduced nitrogen concentrations of both foliage and leaf litter (Wardle *et al.* 2009) and indirectly enhanced total ecosystem carbon storage (Wardle *et al.* 2007). The research is important, firstly because it strongly documents at least one case of ecological 'ripples' of rat invasion, thus expanding our understanding and imaginations of the subtle changes wrought by invading species in all ecosystems. Secondly, as noted by Towns *et al.* (2009), it informs restoration objectives and outcomes because it confirms that restoring below-ground and litter-dwelling invertebrates and associated nutrient cycling after rat eradication first requires successful seabird recolonisation, which may take considerable time.

Increasing 'mainland' restoration

Success with eradications of large suites of species through multiple eradications in fenced sites on the New Zealand mainland has been supported by several factors.

First, the development of pest-proof fencing (Day and MacGibbon 2007), accompanied by refined techniques for eradication and subsequent surveillance (Speedy *et al.* 2007), has enabled multi-species mainland eradications over areas of up to 3400 ha (Maungatautari), with very low levels of pest reinvasion. Mice have proved to be particularly difficult to eradicate, and reinvasion of several species is probably inevitable with time, due to treefalls, water scouring and human error jeopardising fence integrity. For this reason, strictly achieving 'eradication', where immigration is permanently prevented (Bomford and O'Brien 1995), is probably impossible in mainland sanctuaries. The term 'near-eradication' is therefore a better medium-term description.

Unfenced mainland sanctuaries, known as 'Mainland Islands' have achieved many restoration successes (Saunders and Norton 2001; Gillies *et al.* 2003), but also have problems sustaining low residual pest abundance with traps and poisons alone. Even 'near-eradication' is difficult and expensive to sustain without a fence. Fenced sanctuaries routinely target more pest species and eradicate or 'near-eradicate' most of them, promising greater biodiversity gain.

Second, there is burgeoning public interest in community-driven restoration projects. Having learned about dramatic species rescues and habitat repair on offshore islands, many New Zealanders wish to see such restoration in their local mainland landscapes, where species loss has also been profound and is ongoing (Innes *et al.* 2010). This interest has been enhanced by near-shore islands such as Tiritiri Matangi and Kapiti, and mainland sanctuaries such as Karori and Maungatautari. These have offered increased public access to see restored forest communities and threatened species of wildlife previously confined to remote islands. Reflecting this public interest, the total area of managed mainland sanctuaries (defined as targeting at least three major pest species) is now 64,000 ha, cf 37,000 ha of islands free of introduced vertebrates.

Third, islands represent only a small fraction of the total New Zealand environment (Meurk and Blaschke 1990), so that most ecological restoration must focus on the mainland. Alpine and braided river environments, for example, are poorly represented on islands, and so species characteristic of those habitats (eg. birds such as blue duck (*Hymenolaimus malacorhynchos*) and black stilt (*Himantopus novaezelandiae*)) cannot be restored on islands. Meurk and Blaschke (1990) warned that 'predator-free islands… should not be seen as a substitute for mainland protection of representative examples of vegetation-soil systems'.

HOW DO MULTI- AND SINGLE-SPECIES ERADICATIONS DIFFER?

More complex pest eradications

Baits containing the anticoagulant brodifacoum have eradicated or near-eradicated multiple vertebrate species by simultaneously poisoning rodents, hedgehogs, lagomorphs, brushtail possums, and their predators. However, other techniques are often needed to 'mop up' survivors of some target species (Speedy *et al.* 2007). For example, aerial and ground shooting, spotlighting, trained dogs, and trapping are needed for large herbivores such as goats and deer. Techniques required to detect and remove the last few survivors of any taxon tend to be species-specific. Even when generic techniques such as poisoning are employed against, for example ship rats and mice, the types of poisons and bait stations used and their spacing are frequently different for the two species.

The last survivors of other invasive mammals (stoats, ship rats, possums) may be difficult to detect because they are partly arboreal. They thus threaten tree-nesting birds and other fauna such as lizards and large invertebrates within forest canopies. Invertebrate pests such as vespulid wasps can be regarded as 'keystone predators' in New Zealand forests (Towns *et al.* 1997), restructuring forest invertebrate communities and depressing food supplies for native birds (Beggs 2001; Beggs and Rees 1999). Invertebrates remain a difficult problem that requires its own suite of species-specific approaches.

Search and eradication methods targeting plants also tend to be species-specific (West 2002). Because different species occupy different habitats, they require different removal methods, with specific strategies and time-scales to limit subsequent regeneration.

In summary, eradications targeting multiple species demand multiple eradication techniques, require diverse items of equipment, and may call on a wide range of specialist personnel.

Increased benefit:cost ratio per species eradicated

Generally, there is reduced marginal operational cost and increased biodiversity benefit per extra species removed from an island (Overton 2010). Removal of the first species may result in increases in density of remaining pests, or may see remaining pests change diet simply to replace the impact exerted by the first species removed. Both outcomes emphasise the usual result that removal of the final pest provides more biodiversity benefit than removal of the first (Overton 2010).

More stakeholders

In New Zealand, multi-species eradications from near-shore sanctuaries and on the mainland demand much more interaction with stakeholders compared with projects undertaken on remote islands. On remote islands there are no neighbours, and there is usually one landowner and one managing agency. Stakeholders on the mainland include the following:

Volunteers. Many citizens, who live near a sanctuary and share the vision of restoration that project leaders have sparked to practical reality, wish to be actively involved in diverse aspects of sanctuary management. Their skills can be diverse, including engineering, architecture, law and design. In addition, they provide labour for the hours of repetitive physical tasks such as checking fences, traps, bait stations, and tracking tunnels. For example, the Maungatautari Ecological Island Trust based at Cambridge, NZ, has more than 600 volunteers on its books. In 2009 they contributed 67,000 person-hours of labour, which costed at an hourly rate represents the largest single source of 'funding support' for the project.

Corporate sponsors. Sanctuary managers are challenged by the high costs of pest-proof fencing and eradications, plus expenses associated with the capture, housing, disease-screening and translocation of native species to be reintroduced. Corporate agencies are often interested in contributing to such ventures as sponsorship, because the sanctuary offers exposure of the corporate's brand to a large number of visitors. The maintenance and positive political profile of the sanctuary is then in the joint interests of both the local community and the corporate agencies.

Private landowners contributing to the protected estate. Some NZ mainland sanctuaries, composed largely of public land, have incorporated private land by mutual agreement. This is primarily to reduce fencing and lower costs.

Private neighbours. The ways in which sanctuary and neighbouring land management influence each other are still evolving in New Zealand. Killing pests such as feral cats and stoats that range widely can increase the number of prey, such as rabbits, on surrounding farms. Valued domestic cats and dogs that belong to neighbours may stray into sanctuaries and be at risk of injury or death from traps and poisons. Neighbours of fenced sanctuaries have important roles to help maintain the integrity of the fence by careful stock and vehicle management, and to allow access for fence repair, sometimes at night and at short notice, if the fence is breached. Inevitably, wildlife will overflow from the sanctuary to surrounding properties. Such is human nature, this is not always valued; neighbours of Karori fenced sanctuary in urban Wellington, New Zealand, complained that the increased dawn chorus of native birds inside the sanctuary woke them up!

Other users of the estate. While islands are frequently legally reserved as wildlife sanctuaries with limited public access, they are also remote and have few visitors. Mainland and near-mainland sanctuaries differ on both counts, with visitors mostly encouraged. These can include campers, fishers, hikers, boaters, photographers, water skiers, and farmers who have access through the reserve.

Researchers. Easy access to mainland sanctuaries also encourages research from tertiary institutions, since students can cheaply make repeated field visits and have on-site or nearby accommodation. The research questions answerable in mainland sanctuaries are also perhaps more widely relevant to mainland restoration, than questions answerable on remote islands, that tend to have their own unique edaphic-biological characteristics (Meurk and Blaschke 1990), and so are more attractive to researchers.

Paying visitors. The financial sustainability of mainland sanctuaries subject to either continuous pest control, or pest-proof fencing, is uncertain. Getting visitors to pay for access to sanctuaries has been applied at Karori and is perhaps inevitable elsewhere. Cost recovery can result in entry fees beyond the reach of some stakeholders but also raises expectations of the experiences that a paying visitor should receive, or may care to pay for again.

Opponents. People who may oppose a sanctuary undertaking multi-species eradication include ratepayers, if their territorial local authority is a co-funder; anti-poison lobbyists who may contest the humaneness of toxins and traps or dispute evidence about non-target and secondary poisoning risks, and hunters, who value invasive species such as pigs or deer.

WHAT CHALLENGES REMAIN FOR MULTI-SPECIES ERADICATIONS?

We see four challenges for multi-species eradications. These may apply equally to single-species eradications, especially if the latter are undertaken where many people must be involved. These challenges are to: clarify key ecological, social and cultural goals; increase the physical scale of eradications; work with increasing numbers of stakeholders; and build and sustain tactical capacity (such as available bait distribution equipment and skilled helicopter pilots) and knowledge.

The ecological complexities associated with eradications that we have outlined raise particular challenges for managers of sanctuary projects as they attempt to clarify conservation goals shared by all parties. Even when defined, it may be difficult to know if ecological goals are actually being met. Typically, community groups struggle to find the funding necessary to build and maintain pest-proof fences and undertake successful eradications inside them. Detailed monitoring of outcomes and strong research in support can take second place behind these practical realities, so that details of the actual benefits of multi-species eradications become based on perception rather than empirical data. However, empirical data are vital for understanding the order in which multiple pests should be eradicated. Zavaleta (2002) drafted a 'prototype planning guide for averting unexpected eradication outcomes on islands', and Morrison (2010) suggested that risks of unwanted outcomes could be averted by eradicating pests simultaneously or in a 'trophically strategic' order that foresees trophic cascades and considers whether taking out one pest makes another more vulnerable to control.

Given the ecological challenges, defining social and cultural goals seems even more difficult. Yet if multi-species eradications are to take the next logical step in New Zealand they will need to target large areas of mainland – for example, peninsulas where small fence lengths can protect large land areas – or larger islands such as Stewart (174,600 ha) or Great Barrier (28,500 ha). Such areas involve tens of thousands of hectares, may have many residents and considerable vehicular exchange, and face constant risks of reinvasion. Social factors aside, these large areas provide planning, logistical, financial, and monitoring challenges very different from even the largest of the single species eradications on islands. Nonetheless, there are international examples of eradications in rural and urban areas. Respective examples are the successful eradication of mink (*Mustela vison*) from several islands in the Outer Hebrides,Scotland, (Roy *et al.* 2006) and the eradication of coypu (*Myocastor coypus*) from southeast England (Carter and Leonard 2002). Working with resident landowners inevitably imposes constraints on the eradication tools that can be used; some residents may object to the eradications proceeding, and the risks of pests reinvading are undoubtedly higher as residents go about their daily lives, travelling to and from neighbouring places where pests are still present.

Every step of planning and implementing large-scale multi-species eradications demands strong tactical capacity and leadership, details of which should then be captured and communicated. Symposia such as the 2010 *Island Invasives: Eradication and Management Conference* and its 2001 predecessor have been extraordinarily successful at bringing detailed accounts of eradications to publication, and bringing eradication practitioners from around the world together at one place to share experiences and insights.

CONCLUSIONS

We agree with Howald *et al.* (2007) that social acceptance and funding are now more likely to limit future multi-species eradications than say, island area, and that involvement with and agreement of stakeholders, especially local residents, will be essential for sustained success. The success of island eradications to date has undoubtedly been assisted by the remoteness of islands and the fact that many are owned and managed by just one agency focused on conservation outcomes. Both of these factors have freed the eradicating authority to use the most effective tools with minimal need for complex negotiation with diverse other stakeholders. However, on some islands and at many mainland sites, multi-species eradications can be a powerful and effective restoration tool that is greatly needed in many inhabited parts of the world. At mainland sites, multispecies eradication within fenced areas appear to be a particularly promising although complex solution to the reduced roles of indigenous taxa in key ecological processes. So far, no such fenced sites have been built to contain residents whose daily lives must accommodate the fence while protecting native biodiversity from the effects of human activity. In the right social environment even this might be possible.

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Field efficacy of the Curiosity feral cat bait on three Australian islands

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Abstract Predation by feral cats (*Felis catus*) has lead to declines in wildlife populations throughout Australia. Existing tools cannot achieve reductions in cat populations over large areas without presenting a hazard to wildlife. Para-aminopropiophenone (PAPP) formulations are being developed as new tools for the management of feral cat populations. The toxicant formulations are encapsulated within a degradable polymer. The combination of the toxicant formulation and encapsulation provides a robust pellet which is itself implanted inside a moist sausage bait. Pelletised toxicant delivery has been demonstrated to reduce exposure of non-target fauna to bait delivered toxicants. Field evaluations of the bait and pelletised toxicant delivery system have been undertaken at three island sites where the hazard to resident non-target species was minimal. In the first of these trials (April 2008), 6 of 8 radio-collared feral cats died following application of bait at 69 baits km^{-2} over a 60 km^2 area within the French Island National Park (Victoria). In the second trial (April 2009), baits were aerially delivered at 50 baits km^{-2} over a 250 km^2 area on Dirk Hartog Island (Western Australia). Sodium monofluoroacetate (1080) was substituted as the toxicant in this trial as engineering issues prevented production of the PAPP doses. Encapsulated pellets of the non-toxic marker Rhodamine B were implanted into 23% of baits and used as an indicator of cats that would have been expected to have died had PAPP pellets been available. Twelve of 15 radio-collared feral cats died following consumption of bait(s) and of these, nine were positive for Rhodamine. Feral cat activity, monitored over 4 x 10 km transects, indicated a twelve-fold decrease following baiting. For the third trial (August 2009), baits were suspended from purpose-built devices placed at 100 m intervals along the existing road network across an 85 km^2 area within Christmas Island National Park (Indian Ocean). Feral cat activity following baiting was reduced by 87% resulting from consumption of baits by a maximum of 78 feral cats. Further trials are planned for Australian mainland sites to collect efficacy data for purposes of obtaining agricultural chemical registration.

Keywords: *Felis catus*, para-aminopropiophenone, PAPP, efficacy, poison bait, encapsulation, GPS collar

INTRODUCTION

Populations of feral cats (*Felis catus*) first became established in Australia during the mid-1800s (Abbott 2002, 2008) and impact native fauna through direct predation, transmission of disease and competition for resources (Dickman 1996). Feral cats are defined as those animals that live and reproduce in the wild (e.g., forests, woodlands, grasslands, wetlands) and survive by hunting or scavenging with none of their needs satisfied intentionally by humans (Anonymous 1999).

Land managers have used shooting, trapping and/or exclusion fencing to manage feral cat populations in Australia but these techniques have limitations with respect to the area of effective control (Fisher *et al*. 2001). Less labour-intensive techniques, such as poison baiting, has mainly been limited to the arid zone of Western Australia, where surface-laid baits containing sodium monofluoroacetate (1080) is not considered to present a hazard to populations of non-target species due to their higher tolerances to the poison (McIlroy 1981; Algar and Burrows 2004). However, the native fauna of eastern Australia does not have similar tolerances to 1080 (McIlroy 1986; King 1990), which precludes broad-scale baiting for feral cats in these areas. The Australian Government listed the development of an effective toxin and bait for management of feral cat populations as a very high priority (Anonymous 1999; DEWHA 2008).

A collaborative project addresses this requirement through laboratory and field based studies to develop a bait that is humane, target-specific to feral cats and cost-effective. The Western Australian Department of Environment and Conservation (DEC) has previously developed a moist meat sausage bait (*Eradicat*) that consists of kangaroo, chicken fat and flavour enhancers (Algar and Burrows 2004). An automated dosing device injects 4.5 mg 1080 into each bait during production. Baits are air dried and then stored frozen until they are freighted in refrigerated condition to the field. On the morning of use, baits are spread on elevated racks to thaw and 'sweat', a process in which volatile aromatic oils exude from the skin. A residual insecticide (Coopex, Bayer Crop Science, East Hawthorn, Australia) is lightly sprayed over baits which are then bagged and loaded into aircraft or ground-based vehicles for application. This bait has also been found to be attractive and palatable to feral cats at a south-east Australian temperate site (Johnston *et al.* 2007).

Para-aminopropiophenone (PAPP) is a toxicant with improved 'target specificity' based on the reported susceptibility of felids compared to other genera (Savarie *et al.* 1983; Fisher *et al.* 2008). This compound triggers the oxidation of haemoglobin to methaemoglobin, which is unable to transport oxygen (Bright and Marrs 1983). A series of pen trials were conducted to identify suitable PAPP formulations, inclusive of various 'solubilising' excipients and hard impervious coating matrices. An acid soluble polymer encapsulation structure houses the PAPP dose, preventing dispersion of the toxicant into the bait matrix. The hard impervious coating provides a robust toxicant pellet, termed the Hard Shell Delivery Vehicle (HSDV), which is reliably consumed by feral cats and conversely, rejected during feeding by many non-target species (Marks *et al*. 2006; Hetherington *et al*. 2007; Forster 2009; Johnston, *unpublished data*). When utilised with a HSDV, the *Eradicat* bait (without 1080) is known as the Curiosity bait.

In this paper, we describe field efficacy studies of the Curiosity bait, which were required to assist regulatory authorities with registration of the product as an agricultural chemical. We report on trials conducted on three Australian islands in different climatic zones: temperate (French Island – Victoria), semi-arid (Dirk Hartog Island – Western Australia) and tropical (Christmas Island – Indian Ocean). Island studies were the first to be undertaken to

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enable investigation of the efficacy of the Curiosity bait in the absence of canids and domestic cats. There are also fewer extant species of non-target mammals at these sites compared to similar sites on the mainland.

MATERIALS AND METHODS

Site descriptions

French Island (38°21'S, 145°21'E), at 170 km^2, is located in Western Port approximately 70 kilometres south-east of Melbourne. The French Island National Park (FINP) covers 121 km^2 and includes salt marsh, heath, eucalypt woodland and pasture communities (Weir and Heislers 1998; Lacey 2008). Freehold areas on the island are grazed and include residences for permanent and absentee land-owners. Extant native mammal species include long-nosed potoroo (*Potorous tridactylus*), bush rat (*Rattus fuscipes*), swamp rat (*R. lutreolus*), water rat (*Hydromys chrysogaster*) and koala (*Phascolarctos cinereus*). Feral cats established within the National Park from strayed domestic animals and following historical deliberate releases (Lewis 1934). The baiting study was conducted in a 60 km^2 component of the FINP.

Dirk Hartog Island (25°50'S 113°0.5'E), at 620 km^2, is located in Shark Bay approximately 850 km north of Perth. The study was restricted to a 250 km^2 area in the north of the island, in spinifex grassland, low acacia or pittosporum shrub-land (Burbidge and George 1978). The western coast of Dirk Hartog Island (DHI) is rocky in contrast to the eastern coast which is largely sandy. No domestic animals are permitted on the island but there are herds of feral goats (*Capra hircus*) and sheep (*Ovis aries*). Feral cats became established on the island with pastoralists during the late 19th century (Burbidge 2001) and have been implicated in the local extinction of ten mammal species (Algar *et al.* 2011).

Christmas Island (10°29'S 105°38'E), at 135 km^2 is located approximately 2650 km north-west of Perth. This study was conducted along 50 km of existing tracks within the Christmas Island (CI) National Park and adjoining mine lease. Vegetation within the study area consisted of terrace soil evergreen rainforest amongst phosphate mining fields. Approximately 1300 people live in a community on the north-east coast. No ground-dwelling native mammal species remain on the island. However, native land crab species are abundant (Green 1997). A population of feral cats became established following the arrival of settlers in 1888 (Tidemann *et al*.1994).

Baits, poisons and field application

The baits used in these studies resemble chipolata sausages and weigh approximately 15 g when dried. They are manufactured from 70% kangaroo meat mince, 20% chicken fat and 10% digest and flavour enhancers (Patent No. AU 781829) (Algar and Burrows 2004). Approximately 4100, 17000 and 7000 baits were used in the FINP, DHI and CI studies respectively. Baits used for the FINP study were prepared by the authors using domestic sausage manufacturing equipment from meat mince that had been buffered to pH ~8.0. The baits used in the DHI and CI studies were prepared at the DEC bait manufacturing facility and were not pH buffered. A HSDV implanted in one end of the meat baits contained a 78 mg PAPP formulation in the FINP and CI studies. The baits used on DHI were poisoned with 4.5 mg solution of 1080 and 23% of these were implanted with a HSDV containing 30 mg non-toxic Rhodamine B formulation.

Particular site characteristics and other logistical reasons necessitated differences in the method of bait application at each site as described below.

FINP: A Bell Jet Ranger helicopter flying at approximately 20 knots aerially distributed baits at a density of 50 baits km^{-2} (i.e. 5 baits dropped every 10 seconds) on east-west transects spaced at 1000 metre intervals. Baits were also applied around the coast above the high tide line. A total of 3585 baits were dropped from the helicopter and 578 baits were laid along the track network at 100 m intervals from ground based vehicles. A 100 m buffer zone was not baited between the study area and private land. Overall baiting density was 69 baits km^{-2}. Baiting was undertaken on 29 April 2008.

DHI: Radio-collared cats were located on the morning of the baiting day from a single engine light aircraft equipped with VHF telemetry equipment. A flight plan was prepared for the baiting aircraft that consisted of 1 km^2 cells laid over a map of the study site. Baits containing the Rhodamine B HSDV were allocated to the cells where the collared cats had been located. The rest of the site was baited with baits that did not contain the HSDV. A twin engine aircraft flying at 130 knots at 500 feet ASL and guided by an AG-NAV navigation system, was used to drop 16,000 baits in accordance with the plan on 19 April 2009. A timing light indicated when the bombardier was to empty each bag of 50 baits into the drop tube to achieve the desired location and density of 50 baits km^{-2}. Follow-up baiting was undertaken on foot in the vicinity of collared cats that were still alive at >8 days and >13 days after aerial baiting (Johnston *et al.* 2010; Algar *et al*. 2011).

CI: Two baits, tied at the twist link, were suspended from each of 524 Bait Suspension Devices (BSD) (Algar and Brazell 2008) spaced at 100 m intervals along the roadside. A sand pad was formed underneath each device from crushed phosphate dust. Initially, non-toxic baits were provided at each bait suspension device across the site. Fresh baits, each containing a 78 mg PAPP HSDV, were provided following bait removal by a feral cat as evidenced by footprints on the sand pad (Johnston *et al*. 2010; Algar *et al*. 2010). Toxic baits were also supplied at bait suspension devices adjoining these active locations. All baits were replaced every four days to ensure they remained attractive but were not treated with Coopex. Baits were available for the period of 7 - 21 September 2009.

Monitoring

Feral cats were trapped during February 2008 (FINP) and March 2009 (DHI) within the study areas prior to the baiting programmes using padded leghold traps (Victor Softcatch, Woodstream, Pa.; USA). A blended mixture of cat faeces and urine 'Pongo' and a Feline Attracting Phonic (Westcare Industries, Nedlands, Western Australia) were provided at each trap set. Trapped cats were sedated with an intramuscular injection of an estimated 4 mg/kg Zoletil 100 (Virbac, Milperra; Australia). Cats were fitted with GPS datalogger / VHF transmitter collars that included a mortality mode feature and weighed 130 grams (Sirtrack, Havelock North; New Zealand). Radio-collared cats were released at the point of capture and subsequently monitored using an Australis VHF receiver (Titley Electronics, Ballina, Australia) fitted to a handheld yagi antenna or a uni-direction whip antenna fitted to the roof of a vehicle. Monitoring was initiated 14 days after baiting at FINP and two days after baiting on DHI. Transmitters were recovered if they were found to be in 'mortality mode'. Where possible, cause of death was established. PAPP toxicosis in the FINP study was confirmed by assessing presence of bait in the stomach and/or the colour of soft tissues in the mouth. A pale blue colour indicated a deoxygenated condition consistent with PAPP toxicosis. The gastro-intestinal tract was inspected for red staining in the DHI study indicating consumption of the Rhodamine B-HSDV.

Table 1 Morphometrics and fate of collared feral cats at French Island National Park. Baiting was undertaken on the 29th April 2008.

ID	Sex & body weight	Note
0400	Male 3.0 kg	Animal died 29 April. Multiple baits in stomach
1200	Male 3.4 kg	Animal not in baited area. Animal died 24 May following distribution of additional baits on 22 May. Bait in stomach.
1400	Male 3.6 kg	Survived. Animal initially outside baited area. Entered baited area on 1 May.
1600	Male 3.3 kg	Survived. Always within baited area.
1800	Male 3.8 kg	Died 1 March from unknown cause
2400	Female 2.2 kg	Died 11 April from unknown cause.
2600	Female 2.6 kg	Animal died 30 April. Multiple baits in stomach.
3000	Female 2.6 kg	Animal died 29 April. Multiple baits in stomach.
3600	Female 2.8 kg	Data stopped 28 April. Carcass recovered 16 May. Multiple baits in stomach.
3800	Male 3.4 kg	Animal died 29 April. Furred skeleton when recovered on 24 July. Probably died from bait consumption.

Activity monitor plots were constructed at 500 m intervals on the existing track network throughout the study areas using existing soil or suitable substrate transported to the site. Lures were provided at each monitor plot to increase visitation by feral cats, including the Felid Attracting Phonic, feline scent and/or food (pilchard and fried beef liver). These lures were different from those used in the trap sets and were removed when the plots were not being assessed. Cat visitation at the monitoring plots was recorded over five consecutive nights prior to and following baiting to generate a Plot Activity Index (PAI). This index is expressed as the mean number of sand pads visited by the target species per night. The PAI is formed by calculating an overall mean from the daily means (Engeman *et al.* 1998; Engeman 2005). The VARCOMP procedure within the SAS statistical software package produced the variance component estimates. The PAIs before and after baiting were compared using a z-test (Elzinga *et al.* 2001).

As bait station activity on CI could not be ascribed to individual feral cats, a value for the maximum and minimum number of cats poisoned was determined. The total number of toxic baits removed was considered to indicate the maximum number of individuals poisoned. The minimum number of individuals poisoned was calculated by ascribing bait removals from consecutive BSDs to the same animal, even if ten or more stations were involved. The actual number of feral cats poisoned during this programme would be between these two extremes.

RESULTS

French Island National Park

Twelve feral cats were trapped within the study area with collars fitted to six males (3.0 – 3.8 kg) and four females (2.2 – 2.8 kg). Eight of the ten collared cats were known to be alive when baits were distributed. Four cats died as a result of bait consumption as determined by inspection of stomach contents (Table 1). The body of another had deteriorated sufficiently to preclude confirmation of PAPP toxicosis when it was recovered. However, the GPS data for this cat indicated that movement ceased on 29 April (i.e. the day that baits were applied), so it is probable that this cat had also consumed baits. Three collared cats were found alive during the post-baiting monitoring period. One cat was consistently found outside the baited area so 10 additional baits were laid in its vicinity on 22 May. This animal died on 24 May, with PAPP toxicosis confirmed as the cause of death. GPS data indicated that another of the surviving cats was initially outside the baited zone but should have encountered baits on 1 May while the other was always within the baited area. The assessment of feral cat activity on FI at monitor plots conducted prior to and following baiting proved, in this trial, to be inconclusive in the baited area (Table 2). However, an increase in activity at the monitor plots was observed in the unbaited area.

Table 2 Plot Activity Index for feral cat activity at monitor plots (unbaited area n=30, baited area = 102) on French Island.

	Activity Index	Variation	Standard error
Control zone pre-bait	0.080	0.0005	0.023
Control zone post bait	0.280	0.001	0.036
Baited zone pre-bait	0.011	0.00002	0.005
Baited zone post bait	0.009	0.00001	0.004

Dirk Hartog Island

Twenty-one feral cats were trapped within the study area and collars were fitted to 12 males (3.2 – 5.5 kg) and 4 females (3.5 – 3.7 kg). Fifteen collared cats were known to be alive when baits were aerially distributed. Twelve of these cats died after consuming at least one bait; of these, Rhodamine B stain was observed in the gastro-intestinal tracts of nine (Table 3). Three cats were shot as they had not consumed baits by the 1st May (i.e. 12 days after aerial baiting). Two dead uncollared feral cats were located

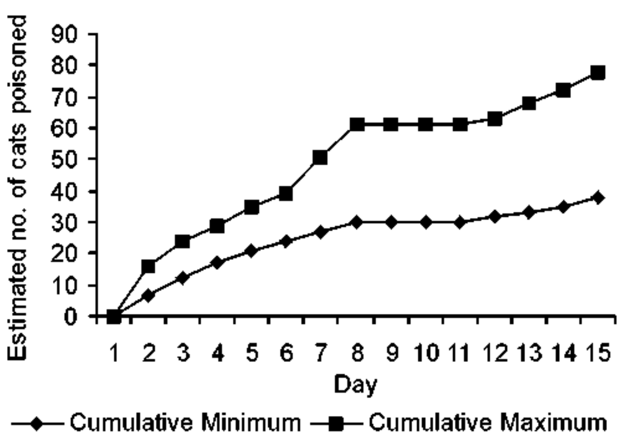

Fig. 1 Cumulative toxic bait removal by feral cats on Christmas Island.

Table 3 Morphometric details and fate of collared feral cats at Dirk Hartog Island. Aerial baiting was undertaken on the 19th April 2009.

Cat ID	Sex & weight	Date & Cause of Death	Comments
DH5	Male 5.1 kg	20 April – Bait	Rhodamine B dye observed
DH5_1	Male 4.2 kg	23 April - Bait	No Rhodamine B dye observed
DH12	Male 5.0 kg	22 April - Bait	Rhodamine B dye observed
DH17	Male 5.0 kg	29 April – Bait	Rhodamine B dye observed
DH27	Male 5.1 kg	2 May – Shotgun	
DH27_2	Male 4.5 kg	20 April – Bait	Rhodamine B dye observed
DH29	Male 4.7 kg	1 May – Shotgun	
B1	Male 3.8 kg	31 March - Unknown	Body recovered 19 April
B2	Female 3.5 kg	27/28 April - Bait	No Rhodamine B dye observed. Bait laid 27 April
B3	Female 3.7 kg	27/28 April – Bait	No Rhodamine B dye observed. Bait laid 27 April
MB2	Male 3.2 kg	20 April – Bait	Rhodamine B dye observed
MB3	Male 3.2 kg	20 April – Bait	Rhodamine B dye observed
MB5	Female 3.7 kg	20 April – Bait	Rhodamine B dye observed
MB6	Male 4.7 kg	20 April – Bait	Rhodamine B dye observed
MB7	Female 3.5 kg	20 April – Bait	Rhodamine B dye observed
MB8	Male 5.5 kg	4 May – Shotgun	

following baiting and both (c. 4 kg male, C. 3.5 kg female) showed Rhodamine B stains in their gastro-intestinal tracts. A comparison of PAIs before and after baiting (Table 4) indicated an 83% reduction in plot activity after the baits were applied ($z = 3.27$, $P < 0.001$).

Christmas Island

Cat visitation was recorded on 96 of the 524 BSDs (18%). Of these, 55 BSDs were visited on more than one night (57%), sometimes multiple times over the baiting period while 41 BSDs were visited only on the one night (43%). Two hundred and sixty-five (3.3%) of the 7860 bait nights accumulated in the study were toxic. A total of 183 baits were removed by feral cats over this period of which 78 (42%) were toxic. The total number of toxic baits removed, and by inference the maximum number of individual feral cats poisoned, was 78 (Fig. 1). The minimum number of individuals poisoned was 38 cats. Feral cats removed non-toxic baits from BSDs without making a return visit when toxic baits were available on 43 occasions.

A comparison of the PAIs before and after baiting (Table 4) indicated an 87% decline in feral cat activity after the baits were spread ($z = 3.17$, $P < 0.001$).

DISCUSSION

Curiosity baits achieved a considerable reduction (or dye-marking) in the feral cat populations and decreased measured cat activity at monitoring plots in all three sites tested. Activity monitoring plots and sandy tracks inspected at FINP indicated low feral cat activity within the study area prior to baiting when compared to the activity in the non-baited area. However, sufficient cats for a statistically robust study were monitored with collars in the baiting area. Data from the GPS dataloggers suggest that baits were consumed by cats within two days of application. Two surviving cats should have encountered bait within three days of application, but no attempt was made to determine whether these cats had consumed bait and survived. It is not possible to determine whether cats consumed aerial or road delivered baits. However, the use of aerial baiting probably led to improved bait acceptance compared with previous studies at this site that only utilised road baiting (Johnston *et al*. 2007). The GPS dataloggers indicated that feral cats made greater use of the dense heathland vegetation than previous studies had indicated (McTier 2000).

Engineering failures prevented manufacture of sufficient HSDVs in time for the DHI trial. Modification of the trial design, which utilised the available stock of Rhodamine B HSDVs in *Eradicat* baits with 1080, provided an adequate alternative to assess the expected efficacy of PAPP. However, conclusions from the results in the DHI study were limited by: i) unavailability of PAPP and ii) only 23% of baits were implanted with Rhodamine B-HSDV's. Nonetheless, *post mortem* examinations after baiting indicated that 80% of the collared cats died following consumption of bait. Changes in feral cat activity on the activity monitor plots following baiting indicates that a significant decrease in activity was achieved. Our data suggest that 10 cats consumed aerially laid baits and 2 were probably poisoned by hand laid baits. Rhodamine B dye was observed in 75% of the cats that died from bait consumption. One collared cat died following consumption of an aerially-delivered bait but did not show any Rhodamine B stains. Unfortunately, as a result of the collar having ceased to collect data, it is not possible to determine whether: i) this cat had moved out of the zone where baits containing the Rhodamine B HSDV had been applied, or ii) it had encountered bait but rejected the HSDV during feeding.

The activity and abundance of land crabs as effective food scavengers on CI required modified baiting procedures (BSDs) to ensure adequate bait availability to feral cats. Previous studies of BSDs demonstrated that they effectively delivered baits to feral cats while minimising access to baits by non-target species (Algar and Brazell 2008). Local land management agencies also required that species removing baits be identified prior to use of a toxic

Table 4 Plot Activity Index (PAI) for feral cat activity at monitor plots on Dirk Hartog Island (n=80) and Christmas Island (n=50).

Site	Time	PAI	Variation	S E
DHI	Pre-baiting	0.078	0.00035	0.019
DHI	Post-baiting	0.013	0.00004	0.006
CI	Pre-baiting	0.060	0.00023	0.015
CI	Post-baiting	0.008	0.00003	0.005

bait which ensured that a minimum number of toxic baits were utilised. More cats may have been poisoned if toxic baits had been provided across the site from the outset, given the 43 instances when a BSD was not revisited. The actual number of feral cats poisoned following consumption of the Curiosity bait was between 38 and 78 individuals (Johnston *et al*. 2010; Algar *et al*. 2010). A more accurate figure cannot be determined given that the identification of individual cats was not possible using the sand pads and that it was likely that some cats visited multiple BSDs prior to the onset of symptoms associated with PAPP toxicosis. Nonetheless feral cat activity at the monitor plots was reduced by 87% during the study demonstrating effectiveness of the Curiosity bait. The very low rate of bait removals (1.3% of available baits) by non-target species could be further reduced using a residual insecticide and larger plates on the BSD (Johnston *et al.* 2010).

OVERALL OUTCOMES

Our data suggest that baits remain palatable and are consumed for at least 10 days after application but consumption was highest the day following bait application. Some feral cats probably consumed multiple baits during each of these trials given: i) the bait density used, and ii) that the first symptoms of PAPP toxicosis only become evident about an hour following bait consumption. If multiple bait consumption is confirmed, baiting density could be reduced or the distribution pattern altered to improve efficacy in terms of cost, ease of application, minimisation of hazard to non-target species or improving probability of bait encounter by all resident feral cats.

These studies demonstrated that feral cat populations can be effectively reduced utilising the Curiosity bait. The applicability of the HSDV for selective delivery of toxic compounds to feral cats has been demonstrated in these studies, but the need to identify and mitigate potential hazards to non-target species from the bait remains a high priority (Marks *et al*. 2006; Hetherington *et al*. 2007; Forster 2009; Johnston unpub. data). In particular, there are no published studies of the sensitivity of PAPP to Australian wildlife other than trials in New Zealand (Fisher *et al*. 2008) on common brushtail possums (*Trichosurus vulpecula*) and Australian magpie (*Gymnorhina tibicen*) (Eason *et al.* 2010). Limited testing has indicated that some other Australian species are highly susceptible (S. Humphreys pers. comm.). For example, species such as large goannas are expected to consume whole baits and are thus unlikely to reject the pellet. In such situations, the strategic timing of baiting operations to periods of reduced foraging activity may assist with: i) minimising bait consumption and ii) increasing bait acceptance by feral cats. Alternatively, different HSDV-toxicant formulations might be developed to which such species are less susceptible. The adoption of an encapsulated pellet prevents dispersion of the toxin throughout the bait medium, which reduces the amount of toxin provided per bait relative to a toxin delivered via an aqueous carrier. Combined with reduced baiting density and/or an altered distribution pattern, there is thus reduced potential for multiple bait encounter by non-target species which diminishes the risk of them ingesting a cumulative toxic dose.

Further trials of the Curiosity bait are planned for Australian mainland sites in the temperate, semi-arid and tropical zone to generate sufficient efficacy data for registration of the bait as an agricultural chemical. A necessary component of these trials will be monitoring and reporting on the impact of baiting operations on populations of non-target species. The use of this bait and toxicant delivery technique may have international application for the management of feral cats or other carnivores. Additionally, the non-toxic Rhodamine B-HSDV can be utilised to provide land managers with a minimally invasive but effective risk assessment tool prior to the conduct of a toxic operation. Unit costs associated with the use of the Curiosity bait will be set by a commercial manufacturer licensed to produce the product.

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Benefits of supporting invasive plant and animal eradication projects with helicopters

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Abstract To eradicate invasive alien species from islands, land managers must have the ability to: detect all individuals, remove all individuals, outpace reproduction, and commit adequate resources to ensure project completion. Any inability to meet these criteria – whether due to technical, financial or political factors – can fate a project to failure. Here, we discuss how helicopter-based methods can increase the likelihood of meeting eradication success criteria, while at the same time increasing effective use of limited resources and enhancing personnel safety. We examine the efficiency and effectiveness of ground-based and aerial-based eradication methods used to eradicate feral pigs and control a suite of invasive plants to zero density on two islands in southern California, USA: Santa Catalina Island and Santa Cruz Island. This study highlights numerous advantages of using an intensive, systematic aerial approach in eradication efforts, as compared to more traditional ground-based methods.

Keywords: Channel Islands, feral pig, *Sus scrofa*, eradicate, effectiveness, systematic approaches

INTRODUCTION

Invasive alien species pose a significant and increasing threat to native biota and unique ecosystems of islands worldwide. Because conservation funding is limited, an imperative for managers is to ensure that threats posed by invasive species are resolved as efficiently and effectively as possible. Eradication of pest taxa often can be a cost-effective strategy relative to alternatives such as perpetual control. But eradications require managers to meet a number of criteria. They must be able to: 1) detect all individuals, 2) remove all individuals, 3) outpace reproduction, and 4) commit adequate resources to ensure completion (Bomford and O'Brian 1995). If these criteria cannot be met, a project risks failure. Fortunately, managers today can review decades of eradication projects. By analysing numerous taxa in a variety of island conditions, it is possible to identify methods that might reduce the risks inherent in eradication efforts, and complete the eradication with greatest efficiency.

In this paper, we compare and contrast the cost and risks of using aerial-based eradication methods versus more traditional, ground-based methods. We do so by examining eradication efforts focused on the invasive feral pig (*Sus scrofa*) and a suite of invasive alien plant (IAP) species on two of the eight Channel Islands off the coast of southern California, USA (Fig. 1). Santa Cruz Island (Santa Cruz), at 249 km^2 is the largest of the Channel Islands; Santa Catalina Island (Catalina), at 194 km^2, is third largest. Both islands have a Mediterranean-type climate, support similar vegetation communities, and exhibit generally similar topographical relief, although Santa Cruz is more diverse due to its larger size and higher elevation (Schoenherr *et al*. 1999). Since the 1800s, each island has experienced a history of intensive livestock grazing that has significantly altered the native ecosystems. Neither island has any native ungulates (Schoenherr *et al*. 1999).

Although both islands share many of the same native and alien taxa, there are important differences. Since 1972, the Catalina Island Conservancy (CIC), a non-profit conservation organisation, has managed 88% of Catalina Island. The remaining 12% is owned by a variety of private land owners. Catalina is the only Channel Island that has an incorporated city (Avalon), with a resident population of approximately 4000 that swells to over 15,000 in the summer months. This has undoubtedly represented a significant challenge to conducting eradications. In addition, the island receives nearly 1.2 million visitors annually (Ann Muscat, CIC President pers. comm.). In contrast, all of Santa Cruz is protected, and the island has relatively few visitors. Santa Cruz is within the Channel Islands National Park (CINP) which owns 24% of the island; the remaining 76% is owned and managed by The Nature Conservancy (TNC), a non-profit conservation organisation. Other than a few management staff, there are no permanent residents on Santa Cruz, although the public does have limited hiking and camping opportunities.

Below, we highlight the similarities and differences between the pest eradication programmes on the two islands. We do so in the context of the aforementioned four eradication criteria, and discuss how the systematic use of a small helicopter can help managers meet those criteria. In contrasting the two projects it is important to note the relationship between the two islands. Channel Islands land managers regularly share lessons learned from conservation activities on other islands. The Santa Cruz feral pig and IAP programmes were thus able to benefit from the prior experience of the Catalina programmes. The Catalina programmes were influenced by invasive species management programmes on other Channel Islands, such as feral sheep (*Ovis aries*) and feral goat (*Capra hircus*) eradications on Santa Cruz and San Clemente islands, respectively.

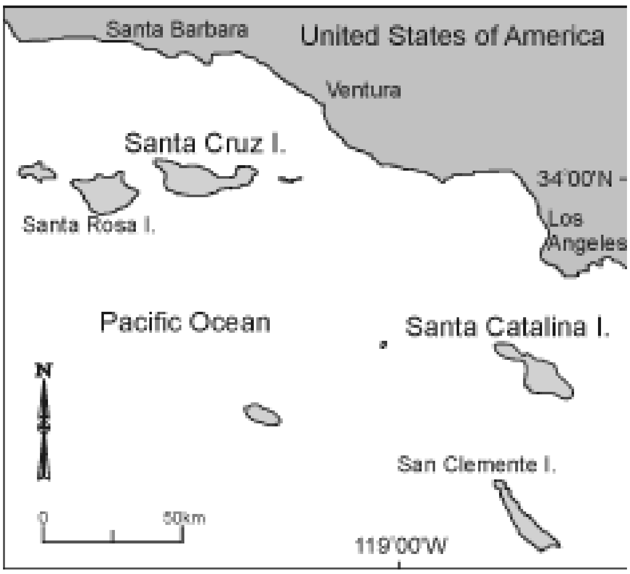

Fig. 1 Santa Catalina and Santa Cruz Islands in the Southern California Bight.

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Table 1 Comparison of feral pig eradication programmes. Data from Macdonald and Walker (2008), Morrison (2007), and Schuyler *et al.* (2002).

Island	Island area (hectares)	Hunting duration (years)	Animals dispatched	Contractor Expense (U.S. dollars)*	Project Completed
Catalina	19,400	10.0	11,855	$3.2 million	No
Santa Cruz	24,000	1.1	5036	$3.9 million	Yes

*See Morrison 2007 for calculations. Fencing costs not included. Adjusted for inflation to 2005 value.

FERAL PIG PROGRAMMES

The attempted eradication of feral pig populations on Catalina Island evolved from a control programme that began in 1990 (Schuyler *et al*. 2002a). Financial constraints, and uncertainty of some CIC board members that eradication was achievable, helped to establish control rather than eradication as the initial goal (Schuyler *et al*. 2002a). Methods and strategies were refined and adapted as the control programme was underway. Throughout the effort, ground hunting with and without dogs, spotlighting, and trapping were used; helicopters were occasionally used to deploy equipment. A helicopter was used as a platform for an aerial shooter only occasionally in the early phases of the project and was later abandoned due to public pressure (Schuyler *et al*. 2002a). In 1998 the objective was changed to eradication, in part because it had become increasingly apparent that sustained control would not accomplish the desired conservation goal (Schuyler *et al*. 2002a). Fencing was then erected to subdivide the island and create hunting zones. Throughout the programme, if hunters encountered multiple pigs they would attempt to dispatch them, even if some were likely to escape (Kevin Ryan and Mark Szydlo, Catalina hunters pers. comm.). As is discussed below, this approach was not used on Santa Cruz.

In contrast to Catalina, on Santa Cruz Island, eradication of pigs was the goal of TNC and CINP from the outset (Morrison 2007). Prior to the beginning of the eradication effort, fenced zones were established across the island. The project was planned and implemented to ensure that the pig populations would remain naive to removal methods as the eradication progressed (Morrison *et al*. 2007). Trapping was employed first, followed by aerial hunting, and only then would Judas pigs and teams of ground hunters with dogs mobilise. By reducing the pig population, the number of pigs encountered by ground hunters was reduced and that increased the likelihood of successful dispatch due to dogs being able to focus on one or two pigs versus many (Macdonald and Walker 2008).

An essential ethic of the hunters on Santa Cruz, whether based on the ground or in the air, was to only attempt to dispatch a pig if: 1) there was very high likelihood of a successful shot, 2) it would be similarly possible to dispatch any other pigs in the vicinity, and 3) it was safe for hunters to do so (Morrison *et al*. 2007; Macdonald and Walker 2008). The skill and discipline required to adhere to this ethic had the additional benefit of reducing the likelihood of injury and escape, which increased the humaneness of the programme (Cowan and Warburton 2011). It was also instrumental in reducing the duration of the project through decreased access time, and overcame the chances of educating pigs to removal methods. This in turn reduced the rate of population replacement and the total number of pigs ultimately dispatched (Table 1). A light piston engine three-person, helicopter (Schweizer 300C) was used to support the full array of activities throughout the project: from aerial shooting to deploying bait and checking traps, from transporting hunters and dogs to tracking Judas pigs and monitoring (Macdonald and Walker 2008).

INVASIVE ALIEN PLANT ERADICATION PROGRAMMES

In 2003, following the near eradication of feral pigs from Catalina, a ground-based island-wide survey for 72 invasive alien plant (IAP) species was commissioned by the CIC. The survey revealed that several species were ideal candidates for eradication based on their limited abundance and distribution (Knapp in press). In 2004, the CIC developed a programme to eliminate 25 species of IAPs from either the CIC property or throughout the island while the infestations were relatively manageable (Knapp in press). Similarly, in 2007, following the successful completion of the Santa Cruz feral pig eradication programme, TNC conducted an island-wide IAP survey for 55 species of IAPs (McKnight *et al*. 2007) and selected 18 species for eradication following the same criteria used on Catalina (Knapp *et al*. 2007). The IAP survey on Santa Cruz was conducted 95% via helicopter and 5% on foot, and covered the entire island (Knapp *et al*. 2009); this is in contrast to Catalina, which was surveyed on foot and only covered a portion of the island. More infestations were mapped on Catalina than Santa Cruz (Table 2). However the species targeted on Santa Cruz had limited distributions, whereas on Catalina some widely established species were also surveyed (Knapp 2004; Knapp *et al*. 2009).

On both islands, surveyors collected the same data on population attributes and delineated infestations in a similar way (Knapp 2004; Knapp *et al*. 2009). Both programmes utilised a similar prioritisation scheme to rank species for management action (Knapp 2004; Knapp *et al*. 2007), and both programmes had a common objective for treatment: control each species to zero density (no above-ground plants remaining), until the soil seed banks are exhausted (Knapp *et al*. 2007; Knapp in press). Currently, both programmes are monitoring seed banks for germination. Both control programmes utilised the same herbicides and used similar application rates and methods (Knapp *et al*. 2007; Knapp in press).

The two efforts differed significantly in: 1) how the surveys were conducted, and 2) how populations were accessed for treatment. Catalina's ground-based survey

Table 2 Comparison of invasive alien plant detection projects. Data from Knapp *et al*. (2009) and Knapp *et al*. in press).

Island	Duration (months)	Populations Mapped	Transects Surveyed (km)	Species	Expense (U.S. dollars)
Catalina	12	32,708	966	72	$35,000*
Santa Cruz	3	5020	4023	55	$161,000

*Survey conducted by a Master's student (Knapp in press).

was conducted primarily by a single graduate student with limited aid from volunteers surveying roads, coastline, major drainages and ridgelines, and scanning the corresponding slopes for infestations (Knapp 2004). The survey on Santa Cruz primarily used a Schweizer 300C helicopter flying within metres of the ground or vegetation; ground survey teams were used only to scan for infestations along roads and highly disturbed sites or sites heavily infested by multiple IAP species (Knapp *et al*. 2009). Since the entire island was surveyed, the Santa Cruz mapping project more closely resembles a census rather than a survey.

Although both IAP control programmes used similar treatment protocols on the infestations, they varied significantly in how the infestations were accessed. All Catalina infestations were accessed on foot by two-person teams. Many infestations took less than an hour to treat, but took nearly the whole day to get to (Knapp, unpublished data). In contrast, on Santa Cruz, applicators were deployed individually by either a Schweizer 300C or 333 turbine engine helicopter to their respective infestations, treating 12 populations on average per day. By eliminating the fatigue associated with accessing infestations or carrying heavy equipment, applicators had more time and energy to scout the surrounding infestation for outlier plants on foot once on the ground, and continued to survey from the helicopter for additional infestations while en route.

DISCUSSION

Eradication Criteria

Each of the following four eradication criteria (Bomford and O'Brian 1995) is dependent on the other three, and the inability to meet any one will adversely affect the overall effort.

1 Ability to detect all individuals

Populations at very low abundance can be exceedingly difficult to detect. Flown at low altitude, helicopters can cover large areas quickly while providing surveyors with an exceptional platform from which to detect eradication targets (McCormick 1999; Welch *et al*. 1999). Many vegetation and topographical features can be scanned with ease from a helicopter. Surveying the same features from the ground can often be labour intensive, hazardous (sometimes impossible), and cost-prohibitive.

2 Ability to remove all individuals

The speed and manoeuvrability of the helicopter increases the ability of the hunter to dispatch groups of animals while ensuring that there is a high probability that no individuals will escape to become educated to eradication methods (Morrison *et al*. 2007). Hunters on Santa Cruz avoided attempting to dispatch pigs if they were not confident that they could dispatch all the animals in the group. Keeping the remaining feral pig population naive to hunters and helicopters was the key factor of the success of the Santa Cruz pig eradication (Macdonald and Walker 2008) and may be the main factor why Catalina is still not free of pigs (Morrison 2007) (Table 1).

Incipient IAP infestations are relatively small and quick to remove, but access time can be considerable (Table 2). For example, a single *Cortaderia selloana* (pampas grass) plant can be treated with herbicide in approximately five minutes; but remote infestations may take hours to access on foot. In addition to reducing access time, the helicopter provides a vantage for another rapid survey of the area surrounding the infestation.

3 Ability to outpace reproduction

The mobility and speed of a helicopter reduces access time, which enables the eradication team to outpace the reproduction of the target population. Ground-based access and detection methods can be restricted by road conditions, moving populations, and other limitations (Table 2).

4 Ability to commit to completion

Land managers often struggle to maintain the resources for a consistent level of staffing, equipment, and funding. Eradication projects can also be delayed due to political and social pressure (Temple 1990), which can jeopardise progress made towards completing the eradication (Morrison *et al*. 2011). For example, animal rights activists attempted to halt the feral pig eradication on Santa Cruz Island through multiple legal actions, and forced the CIC to adopt more costly removal methods during the last months of the goat eradication programme (Schuyler *et al*. 2002b). Rapid completion of the project reduced the exposure of the project to such potential disruption, and so was an important means of reducing the risk of failure.

Indirect Expenses of a Longer Project

The costs of inefficiencies in eradication programmes are many and varied: 1) the physical and emotional well-being of personnel; 2) impacts to habitat due to "bush whacking" (including dispersal of invasive taxa, soil disturbance, vegetation damage, and wildlife disturbance); 3) prolonged input of pesticides into the environment; 4) indirect monetary costs associated with managing and housing contractors; 5) expended political capital with regulatory agencies, funders, local community members and supporters stemming from disagreement with the projects objective, lawsuits, and negative press; and 6) opportunity costs of sustaining focus on one project at the expense of other priorities. These expenses are rarely (if ever) tracked, but are considerable and can have long-lasting repercussions.

Personnel that see progress being made, and are not fatigued, have a better chance of detecting and responding to an eradication target, and bringing an eradication programme to completion. In contrast, the health, stamina, and morale of project personnel can suffer as a project wears on – with risk of injury increasing in a negative feedback cycle. Retention of personnel becomes much more difficult when eradication objectives are not reached quickly. The emotional toll of an eradication attempt can be tremendous. For example, a Catalina pig hunter expressed how he felt traumatised by four years of dispatching animals with no end in sight (Anonymous pers. comm.).

The CIC lost several of its local volunteers who disagreed with pig eradication; some became vocal opponents of the project in the local community. Even CIC personnel not involved with the pig eradication were regularly accosted outside of the workplace by members of the local community. A divide developed within the organisation between staff that supported the project and those that did not, and this disagreement overshadowed daily operations. A shorter programme may not have swayed opposition against the eradication, but a protracted programme kept it at the forefront.

Helicopter Use

It may seem obvious that the use of a helicopter to eradicate invasive taxa will help meet eradication criteria and speed up eradication projects. Helicopters are not a new tool to conservation (McCormick 1999; Schuyler *et al*. 2002a). Why, then, would managers opt not to utilise helicopters in their projects? Helicopters are not free from stereotypes, including that they are dangerous and costly. Regardless of whether the result was the same, a helicopter accident would likely be more spectacular than an accident on foot or by an automobile, and for this reason helicopters

Table 3 Comparison of invasive alien plant eradication programmes. Data from Knapp (2009) and Knapp (in press).

Island	Injuries treated in hospital*	Populations targeted	Area treated (hectares)	Treatment months	Species	Expense (U.S. $)
Catalina	7	404	11	24	25	$1,000,063
Santa Cruz	0	421	7	7	18	$520,000

* Knapp, unpublished records.

may seem more dangerous. This is not to suggest that danger is not associated with helicopter flying at low altitude in rugged terrain, but risks associated with ground-based activities are often overlooked. And although land managers may be at first daunted by a helicopter's hourly rate, considerable saving can accrue by the reduction in access time afforded by helicopter use (Table 3).

Aerial shooting of vertebrates can also be perceived as inhumane. Yet, due to the speed and manoeuvrability of a helicopter it is arguably more humane for an expert aerial shooter to dispatch an animal than it is from the ground. Like all tools, there are various helicopter models that are more suitable to this task than others. The Schweizer 300C and 333 helicopters, flown on Santa Cruz, each have attributes which made them ideal choices for the work described here. The biggest advantage of these machines was their reliability and cost-effectiveness to operate, which enabled them to be flown when they were needed (Macdonald and Walker 2008).

A tool is only as effective as its user. Pilots operating a helicopter must be experienced and able to safely deploy eradication personnel and their equipment in rugged terrain, often under high wind or other adverse weather conditions. A pilot working on an eradication project must also be able to detect the target of the eradication effort. In the case of supporting projects focused on large vertebrates, the pilot ideally is also a skilled hunter, with an understanding not only of the behaviour of the target but also the requirements of the shooter, so as to be able to position the helicopter optimally.

CONCLUSION

By planning helicopter support as an integral component of an eradication strategy, land managers can increase the likelihood of the project success. The right helicopter piloted by an experienced pilot can be a safe, humane, and cost-effective means to eradicate myriad pest taxa. Regardless of the eradication task, enhancing detection and reducing access time is vital to achieve an eradication goal, thus freeing land managers to direct limited funds to other conservation priorities.

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Planning for the eradication of feral cats on Guadalupe Island, México: home range, diet, and bait acceptance

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Abstract Feral cats (*Felis catus*) introduced to new environments have caused the extinction of many vertebrate species, including six species of birds on Guadalupe Island, México. To save species from extinction and restore natural processes, cats have been eradicated from islands using a variety of techniques. Eradication campaigns have to be planned carefully; ideally supported by information about the population to be eradicated. Our study focuses on home range estimation (fixed kernel); bait consumption by feral cats and non-target species; and diet of feral cats on Guadalupe Island. Home range was 76 to 1098 ha (KE 95) and core areas 21 to 196 ha (KE 50). Feral cats and non-target species including Guadalupe junco (*Junco hyemalis insularis*), Guadalupe rock wren (*Salpinctes obsoletus guadalupensis*), western gull (*Larus occidentalis*), and house mouse (*Mus musculus*) consumed baits. Items most commonly found in diet samples were mice (66.5%) and birds (16.8%). Male cats were 2.9 ± 0.6 kg, and females 2.4 ± 0.9 kg. The results of this study will inform eradication decisions for Guadalupe Island, especially regarding the use of poison baits.

Keywords: Morphometrics, *Mus musculus*, *Junco hyemalis insularis, Salpinctes obsoletus guadalupensis, Larus occidentalis,* non-target species, birds, poison baits.

INTRODUCTION

Global extinctions recorded over the past six centuries have been dominated by insular species, and introduced mammals are recognised as the main cause (MacPhee and Flemming 1999; Aguirre *et al*. 2005). Since 1600, 27% of mammal extinctions in the world have been on oceanic and oceanic-like islands; 28 reptile taxa have become extinct (Honegger 1981; Alcover *et al.* 1998) and 90% of bird extinctions have been insular forms (Johnson and Statterfield 1990). The probability of extinction is 40 times higher for an insular species than for continental species (Johnson and Statterfield 1990). One of the most damaging introduced species on islands is feral cats (*Felis catus*), which have been responsible for numerous extinctions worldwide (Iverson 1978; Jehl and Parks 1983; Mellink 1992; Veitch 2001; Tershy *et al*. 2002). For example, on Mexican islands, cats are thought to be responsible for the extinction of at least 16 taxa of birds and mammals (Aguirre *et al.* 2011).

Birds that evolve in predator-free environments often lack defences against new species (Whitaker 1998; Blackburn *et al.* 2004), and rapidly succumb to pressure from predators such as cats. This could include a combination of lack of predator awareness behaviour and habits that make them more vulnerable, such as feeding and nesting on the ground (Simberloff 1995). Cats were introduced to Guadalupe Island, México, in 1885 as an attempt to control the house mouse (*Mus musculus)* introduced ten years earlier (Moran 1996). Instead, the cats exterminated six species of endemic birds: Guadalupe ruby-crowned kinglet (*Regulus calendula obscurus*), Guadalupe Bewick's wren (*Thryomanes bewickii brevicauda*), Guadalupe rufous-sided towhee (*Pipilo maculatus consobrinus*), Guadalupe northern flicker (*Colaptes auratus rufipileus*), Guadalupe caracara (*Caracara lutosus*) and the Guadalupe storm-petrel (*Oceanodroma macrodactyla*). The first extinction was just seven years after the cats were introduced (Jehl and Everett 1985). Cats also extirpated the red-breasted nuthatch (*Sitta canadensis*), white-throated swift (*Aeronautes saxatalis*), red crossbill (*Loxia curvirostra*) and red-tailed hawk (*Buteo jamaicensis*), as well as caused a decline of several populations of seabird species such as Xantus's murrelet (*Synthliboramphus hypoleucus hypoleucus*) and Cassin´s aucklet (*Ptychoramphus aleuticus*) (Keitt *et al.* 2005).

Globally, feral cats have been eradicated from 75 islands (Campbell *et al.* 2011). The techniques used have evolved from the more traditional such as trapping, shooting, and the use of hunting dogs, to the more sophisticated such as special delivery methods for poisons (Marks *et al.* 2006). Eradication requires careful planning, selection of techniques most appropriate to the site, and relevant knowledge of the ecology of the target species (Bonnaud *et al.* 2011). Eradication campaigns against cats need support from research on movements and bait acceptance so existing techniques can be improved (Nogales *et al.* 2004). Information that now informs decisions about how and when to implement eradications includes studies of the diet of feral cats on islands (Bonnaud *et al.* 2011), their home ranges (Smucker *et al.* 2000; Edwards *et al.* 2001; Molsher *et al.* 2005), and bait acceptance (e.g., Wickstrom *et al.* 1999; Algar *et al.* 2007).

Diet studies for feral cats have often tried to quantify the impact of cats on native species (Paltridge *et al.* 1997; Bonnaud *et al.* 2011). Such studies can also highlight the relevance of particular prey to the eradication campaign and thus the likely effectiveness of the eradication attempt (Fitzgerald *et al.* 1991). For example, eradications may be most likely to succeed when the main prey species is scarce (Veitch 1985). If cats are to be eradicated from Guadalupe Island, answers are required for two main questions: 1) which are the most common prey species, and 2) how do populations of these species fluctuate throughout the year? Studies aimed at answering these questions will also generate new information regarding cat diet on Mexican islands. So far, cat diet analyses are only available for two islands in the country (Arnaud *et al.* 1993; Espinosa-Gayosso and Álvarez-Castañeda 2006).

Baits used to attract cats to traps or poisons can vary in effectiveness (Wickstrom *et al.* 1999). In addition to the diet of cats on Guadalupe, we analysed the acceptance of baits successfully used elsewhere for feral cats and non-target species. Although poisoning of some non-target species may be unavoidable during an eradication, there may be ways that these effects can be minimised (Veitch 1985). The first step is to determine which species are potential non-targets.

We also investigated home range characteristics of feral cats, which can inform decisions about the optimum spacing of baits or traps (Edwards *et al.* 2001). Existing home range studies on cats show great variation between habitats and locations (islands or mainland) (Edwards *et al.* 2001; Harper 2004; Molsher *et al.* 2005; Schmidt *et al.* 2007).

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MATERIALS AND METHODS

Site Description

Guadalupe Island is 24,171 ha, rises to 1298 m, and is 260 km off Baja California Peninsula, México (Fig. 1). The island's climate is influenced by the cold California Current and characterised by wind, fog, and winter rainfall (León de la Luz *et al*. 2003). Average temperature is 17.2°C (Hastings and Humphrey 1969) and annual rainfall is 250 mm (Castro *et al*. 2005). The main island, islets and surrounding waters are included in the Guadalupe Island Biosphere Reserve, administered by the Mexican Federal Government's National Commission of Natural Protected Areas.

In total, Guadalupe has 139 species of birds (Quintana-Barrios *et al*. 2006), including 10 species of breeding seabirds (Luna Mendoza *et al*. 2005). The invertebrate fauna is very diverse, including 11 species of endemic land snails. There are no native amphibians, reptiles, or terrestrial mammals. Colonies of northern elephant seal (*Mirounga angustirostris)*, Guadalupe fur seal (*Arctocephalus townsendi*) and California sea lion (*Zalophus californianus*) are present (Moran 1996). After its discovery in 1602 (Moran 1996), sealers and goat hunters visited the island until the 20th century. The Mexican Navy and Local fishermen established permanent settlements on the island in the 19th and 20th centuries respectively.

Guadalupe has 223 plant species, including 39 that are endemic (Rebman *et al.* 2005; Junak *et al*. 2005). Pine, cypress, and palm forests, oak and juniper woodlands, as well as chaparral, grassland, sage scrub, and maritime desert scrub were the major habitat types before goat introduction (Oberbauer 2005). Now, only 6% of the forest remains, the chaparral no longer exists, and the grassland has increased from 1250 ha to 12,800 ha (Oberbauer 2005), due to grazing by feral goats and the introduction of weeds. The only remaining pristine habitat is scrub vegetation on the islets, which never had goats or other exotic mammals. As part of a restoration project, goat and dog eradication started in 2002. Dogs were eradicated in 2005 and the last Judas goats were removed in 2010 (Julio Hernández-Montoya pers. comm.). The only remaining introduced mammals are cats and house mice.

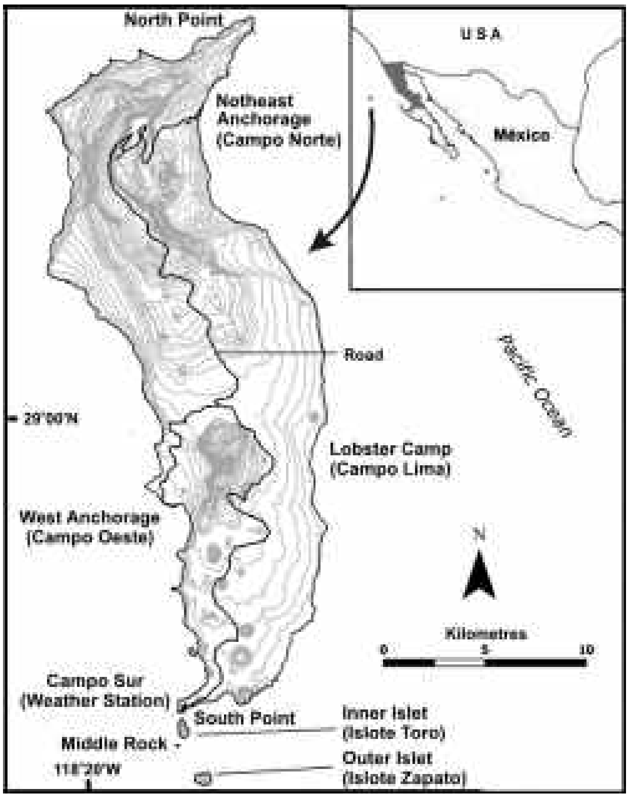

Fig. 1 Guadalupe Island. Location and significant features.

Feral cat population and biology

Home range

Estimates of home range size for feral cats were conducted from May to October 2009. Victor Oneida Soft Catch leg-hold traps (No. 1.5 Oneida Victor Inc. Ltd., USA) were set on trails or in caves (Veitch 1985; Wood *et al.* 2002) using fried fish, fried canned tuna or sardine as bait. Trapped cats were anaesthetised using 0.2-0.4 ml of 5-10 mg/kg zolazepam and tiletamine (Zoletil, Virbac) given intramuscularly (Virbac 2009) and fitted with mortality-sensitive VHF transmitters (Model TXE-311C, 31 gr, 163.499 – 163.959 Mhz, Telenax MX). Morphological attributes such as weight, sex, and age, were measured. Collared cats were released near their capture location and monitored daily using a Yagi folding antenna and a portable receiver (Model WTI-1000, Wildlife Track Inc. USA). Position, time of day, and bearing were recorded (Harper 2004; Molsher *et al*. 2005). Triangulation (Kenward 2001) was used to determinate approximate locations of collared cats. These data were then processed in software Locate III (Pacer Computing, Tatamagouche, NS, Canada). Cat positions were calculated with 95% confidence and incorporated into a Geographic Information System using ArcGis 9.2 (ESRI Inc., Leica Geosystems GIS Mapping, Microsoft Corporation, LizardTech Inc. and Independent JPEG Group) and displayed on a Quickbird image (DigitalGlobe Inc. USA) of the island. Home Range Tools (HRT) for ArcGis (Rodgers *et al*. 2007) were used to estimate the home ranges of feral cats. The Kernel Density Estimation (KDE) method was used, as recommended by Laver and Kelly (2008). Kernel (KE) 95% was used to estimate home range. Core area was calculated using KE 50%. The fixed kernel smoothing parameter was used (Edwards *et al.* 2001; Kenward 2001). Home ranges were calculated with ≥20 locations for each individual and core area with ≥10 locations (Harper 2004; Molsher *et al*. 2005).

Baits

Beef and chicken baits were made by local manufacturers, following the specifications for Eradicat developed by the Department of Environment and Conservation of Western Australia (Algar *et al.* 2002; 2007). Baits contained 80% meat and 20% fat with monosodium glutamate as a flavour enhancer. The baits were 60-70 mm long x 10-15mm diameter and 20g dry weight.

Bait take by feral cats and non-target species was evaluated. The major species of concern were the endemic Guadalupe junco (*Junco hyemalis insularis*), Guadalupe rock wren (*Salpinctes obsoletus guadalupensis*) and Guadalupe house finch (*Carpodacus mexicanus amplus*) as well as the native burrowing owl (*Athene cunicularia*), American kestrel (*Falco sparverius*), western gull (*Larus occidentalis*), mourning dove (*Zenaida macroura*) and western meadowlark (*Sturnella neglecta*). Bait take by house mice was also evaluated.

Bait uptake trails were established in cypress forest, scrubland, grassland, and on the coast. The habitats used were to enable different species to be targeted rather than for comparing habitats. In each habitat, three transects were established 200m apart with eight sand plots (stations) 100m apart along each (a total of 96 stations). Each station was cleared of vegetation and a 1m diameter of sifted dirt or sand was laid to record all animal tracks (Linhart and Knowlton 1975). One bait was placed in the middle of each station, alternating between beef and chicken (Kavanaugh and Linhart 2000). The sand plots were surveyed between

21 October and 13 November 2009 on three consecutive days. After animal tracks were recorded, the stations were then reset by raking over tracks and replacement of baits. Four cameras (Model Trophy Cam, Bushnell Corporation, USA) were used to record bait consumption; two placed on stations with beef sausages and two on stations with chicken sausage. Cameras were installed 30-50cm above ground in front of the stations (S. Robinson pers. comm.), set to record 20-30 seconds of video with a minimum interval of three minutes between recordings, and set for one night on each transect.

Bait consumption was also tested on 24 feral cats held in cages (160x110x110 cm) for four and seven days during October-November 2009. Cats were fed each day at the same hour with fresh meat. During the last day of captivity, three beef and three chicken baits were placed in the cages. Preference of consumption was recorded by direct observation and by the cameras (Marks *et al*. 2006), which were set to take videos every 20-30 seconds with a minimum interval of 10 seconds between shots (Clapperton *et al*. 1994).

Morphological attributes

Between June and December 2009, feral cats were captured at several places on the island using Victor Oneida Soft Catch leg-hold traps and Tomahawk Live Traps (Model 207, 81.3 x 25.4 x 30.5 cm, Tomahawk Live Trap Co. USA) baited as described above. Cats were anaesthetised using procedures described above and euthanized with a heart lethal injection using 0.5-1.0 ml of sodium chloride (Kelefusin, PiSA), at a dose of 40-70 mg/kg (Phillips *et al*. 2005; AVMA 2007). The sex and age (juvenile or adult by tooth wear following Logan *et al*. 1986), coat colour, weight (± 100g), and head-body length (± 10mm) were recorded.

Diet

Stomach contents and scats collected from cats captured were analysed and separated into four categories: house mice, birds, insects or plant material. Frequency of occurrence and relative frequency were calculated for each diet sample. Frequency of occurrence of each category was calculated by dividing the number of diet samples containing each category by the total number of diet samples analysed. Relative frequency was calculated by dividing the frequency of occurrence of each prey item by the total of frequencies of occurrence for all prey items (Smucker *et al.* 2000).

RESULTS

Feral cat population and biology

Home range

In total, 17 cats were caught over 129 trap-nights and transmitters deployed on 12 males (11 adults and 1 juvenile) and 5 females (2 adults and 3 juveniles). Of the cats with transmitters, eight were located more than 20 times over 2100 hours of tracking (Table 1). The average home ranges were 510.4 ± 353.8 ha for males and 226.5 ± 118.1 ha for females (Table 2).

Baits

On transects, 69.16% of the baits were consumed. There was no significant preference between beef and chicken baits ($t = -1.844$, $df = 8.79$, $P > 0.05$; Table 3).

Of the stations where baits were consumed, 28% had images showing the process of consumption. At stations where there was a combination of sign on the raked sand and images obtained from camera traps, there were visits by cats, Guadalupe rock wren, Guadalupe junco, western gull, and mice. Burrowing owls visited the stations but showed no interest in the baits. Tracks or images of Guadalupe house finch, American kestrel, mourning dove, and western meadowlark were not detected at the stations.

Of 24 cats held in captivity, 22 (91.7%) consumed at least one bait and 75% consumed at least three of the six baits offered. Chicken bait was preferred (62.5%) over beef bait (29.17%).

Morphological attributes

In total, 278 feral cats were captured (3548 trap-nights). The coat colour was 77.4% tabby, 21.4% black and 1.2% black and white (Table 4).

Diet

In total, 140 diet samples were analysed, 14.3% were from summer and 85.7% from autumn (Table 5).

The bird species most commonly found in diet samples were mourning dove, Leach's storm petrel (*Oceanodroma leucorhoa*) and Guadalupe Junco, but there was no further analysis of their relative contributions.

Table 1 Home ranges and core areas (ha) for collared cats. Age = (A) adult; (J) juvenile.

Cat No	Age	Sex	No. places	Home range (ha) KE 95	Core area (ha) KE 50
G01	A	M	28	186	33
G02	A	M	25	495	105
G05	A	F	25	310	69
G09	A	F	31	143	27
G11	A	M	25	485	76
G12	J	F	26	76	21
G14	A	M	20	288	60
G17	A	M	20	1098	196

Table 2 Home ranges of adult feral cats on Guadalupe Island and other locations.

Location	Sex	n	Home range (ha) KE 95	Core area (ha) KE 50
Guadalupe Island [1]	M	5	510.4 ± 353.8	94 ± 62.7
	F	2	226.5 ± 118.1	48 ± 29.6
Australia [2]	M	3	103.1 ± 91.9	18.6 ± 13.9
New South Wales [3]	M	11	25-575	7-152
	F	4	126-310	11-68
Stewart Island [4]	M	8	1815 ± 360.3	
	F	3	1065 ± 241.6	

[1]This study; [2]Edwards *et al.* 2001; [3]Molsher *et al.* 2005; [4]Harper 2004.

Table 3 Bait consumption by feral cats and non-target species.

Species	Bait	Consumption (%)
Cat	Chicken	78.21
	Beef	63.29
House mouse	Chicken	10.26
	Beef	25.32
Guadalupe rock wren	Chicken	5.13
	Beef	6.33
Guadalupe junco	Chicken	1.28
	Beef	1.27
Western gull	Chicken	5.13
	Beef	3.80

Table 4 Measurements of feral cats on Guadalupe and other islands.

Island	Sex	Weight (kg)	Max. (kg)	n	Head and body length (mm)	Max. (mm)	n
Guadalupe[1]	M	2.87 ± 0.58	4.6	141	489 ± 36.0	550	91
	F	2.35 ± 0.94	3.5	52	465 ± 28.8	530	34
Little Barrier[2]	M	2.95	4.1	18	473	530	21
	F	2.23	3.8	35	440	320	40
Cocos Islands[3]	M	3.38 ± 0.07	4.8	63			
	F	2.69 ± 0.06	3.7	76			
Macquarie[4]	M	4.3 ± 0.06	5.5	74			
	F	3.7 ± 0.09	5.8	54			

[1]This study; [2]Veitch 2001; [3]Algar *et al.* 2003; [4]Brothers *et al.* 1985.

DISCUSSION

In this study, our main interest was to assess the cats' minimum home range so that any eradication programme using baits would spread them at a density accessible to every cat. On Guadalupe, as in other studies, females had the smallest home ranges (Table 2). Home ranges on Guadalupe and Stewart Island (Harper 2004) were larger than those found on mainland Australia (Edwards *et al.* 2001; Molsher *et al.* 2005). The Guadalupe study was done during autumn, when food resources were abundant. During winter, which will be the best timing for eradication, food resources will be scarcer and in consequence we expect home ranges to be larger.

The size and weight of feral cats on Guadalupe are similar to those reported by other studies (Table 4). Considering the home ranges and morphometrics of cats on this island, we believe that eradication of feral cats from Guadalupe Island is possible using aerial broadcast poison baits at a rate of 100/km^2 to knock down the population (Algar *et al.* 2001, 2002), followed by a rapid response using traditional techniques of trapping, shooting, and hunting dogs.

Bait trails showed that baits similar in size and characteristics to Eradicat could work on Guadalupe Island. Eradicat cannot be imported to Mexico but a similar product can be manufactured. As we expected, bait consumption by house mice was high (Table 3). Interference by house mice will likely be less in winter but will have to be considered for the eradication.

Baits were consumed by three species of birds, which may result in non-target poisoning. Bait consumption by Guadalupe junco and western gull was expected but not the consumption by Guadalupe rock wren. Junco and rock wren are endemic species. The western gull is the only gull breeding on the island and may be an endemic race (Hubs 1960 cited in Jehl and Everett 1985). Mitigation measures required for these species, and further assessment for each non-target species, will have to be included in the eradication planning process. Potential mitigation techniques include the capture and temporary holding of non-target bird species (Howald *et al.* 2003, 2010) or development of encapsulated poison within baits that are unable to be consumed by these non-target species (Marks *et al.* 2006; Hetherington *et al.* 2007). These mitigation actions will require further testing and validation on site. Because the use of 1080 is banned in México, new toxins may need to be evaluated for use on Guadalupe. For example, para-aminopropiophenone (PAPP) may be suitable for the eradication of feral cats (Johnston *et al.* 2010; Eason *et al.* 2010).

Table 5 Cat diet. Frequency of occurrence and relative frequency of prey.

		Mice	Birds	Insects	Plants
Summer (n=20)	Freq of occurrence	63.6%	31.8%	--	4.5%
	Relative frequency	70.0%	35.0%	--	5.0%
Autumn (n=120)	Freq of occurrence	65.4%	15.4%	4.3%	14.8%
	Relative frequency	88.3%	20.8%	5.8%	20.0%

Cats held in captivity consumed chicken and beef flavoured baits but showed a preference for chicken, perhaps because of their higher fat content compared with beef baits. Nevertheless, chicken baits are more difficult to preserve and store than those made of beef, which limits the use of chicken baits in the field. Since beef baits were also accepted and consumed, particularly in the field trials, these baits should be adequate for a cat eradication programme. Other baits could be tested particularly those with at least some chicken or fish to enhance the odour attraction. Fish baits could be considered in the future, but some studies have suggested that they are less reliable for use in the field (Wickstrom *et al.* 1999).

On Guadalupe Island, house mice predominated in the cats' diet (64.4%) followed by birds (23.6%), and plant material (9.7%) (Table 5). Insects were only present during autumn (4.3%), but this could be due to a larger sampling effort. In summer, cats consumed almost exclusively mice and birds. In autumn, the percentage of bird consumption was lower and higher for plant material. The relative abundance of cats on Guadalupe declines during winter, which coincides with the collapse of the house mice population (Luna-Mendoza *et al.* unpubl. data) and the absence of seabirds. Eradication should thus be most effective in winter because the mouse population is probably regulating the abundance of cats. Seasonal or yearly mouse plagues have been reported by locals as the mice seem to be regulated by food availability after rain, when numbers increase, followed by population collapse during winter. It is also possible that vegetation changes after goat eradication are influencing mouse abundance. The seabird population on Guadalupe is seasonal and not large enough to sustain a large cat population.

Questions remain regarding the potential effects of cat eradication on the mouse population. Conceivably, there is potential for mesopredator (mouse) release, which could be more damaging to the natural value of the islands than the current impact of cats. Some studies suggest that the removal of cats (superpredators) increase mesopredator communities such as rats (*Rattus sp.*), which can then cause more damage to prey populations (Russell *et al.* 2009). The negative impacts of house mice on birds are much less known than the effects of rats, but some studies (Wanless *et al*. 2007; Jones and Ryan 2009), suggest that mice could be a serious threat for seabirds. In contrast, Blackwell *et al*. (2003) suggests that ship rats (*R. rattus*) and house mice seem to be regulated more by food availability than by predator pressure. Under this scenario, the eradication of feral cats in Guadalupe might not affect the house mouse population. However, because the effects of house mouse eruptions due to cat removal are difficult to predict, the simultaneous eradication of house mice and cats should be considered.

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A successful mouse eradication explained by site-specific population data

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Abstract Invasive rodents have been responsible for the extinction of many species on islands. House mouse (*Mus musculus)* eradication attempts have been less successful than introduced rat (*Rattus* spp.) eradication attempts and research is needed to identify the reasons for this disparity. We studied, and successfully eradicated, a mouse population on a small (6 ha) island in northern New Zealand in an attempt to characterise possible behavioural factors influencing eradication outcome. We monitored pre-eradication mouse movements with radio-tracking and trapping to provide guidance on grid-spacing for bait stations, which are a common tool used in rodent eradication and reinvasion monitoring protocols. Mouse densities on the island were estimated during three capture-mark-recapture (CMR) sessions in January, March and May 2008. Mice were then trapped almost to extinction in August 2008 and poison baits were used to eradicate the survivors. Removal trapping data combined with WaxTag interference rates provided a final density estimate of mice in winter (August in New Zealand), the period when most eradications are attempted. Densities on the island ranged from 8.8-19.2 mice/ha, with home ranges varying from 0.15-0.48 ha. Eradication success was monitored intensively using tracking tunnels and WaxTags and was confirmed in December 2008 using a trained rodent monitoring dog. Information gathered during this study can be used to make recommendations to improve the success of future mouse eradication attempts. One of the key recommendations is to identify areas of complex habitat (such as dense ground cover) where mice may not come into contact with poison and adjust eradication methods to specifically target such areas.

Keywords: House mouse, *Mus musculus*, density estimate, home range estimate

INTRODUCTION

The house mouse (Rodentia: *Mus musculus*) became commensal early in human history (Cucchi and Vigne 2006), was then widely spread by human activity (Cucchi 2008; Searle *et al.* 2009), and is now one of the most widely distributed mammal species (Rowe 1973; Pocock *et al*. 2005). House mice (hereafter: mice) spread disease (Langton *et al.* 2001), consume cultivated crops (Stenseth *et al.* 2003), and prey on native fauna such as birds, lizards, and invertebrates (Howald *et al*. 2007; St Clair 2011). Some of the worst impacts of mice on native ecosystems are seen on islands where native fauna and flora evolved without mammals (Diamond 1989; Angel *et al.* 2009).

There have been numerous attempts to eradicate mice. However, the global failure rate for these attempts on islands is 38% (MacKay *et al.* 2007), compared with only 5% for Norway rats (*Rattus norvegicus*) and 8% for ship rats (*R. rattus*) (Howald *et al.* 2007). These failures raise the question: why are mice harder to eradicate than rats? Our study was designed to investigate some of the possible behavioural reasons for these failed eradications.

New Zealand is an oceanic archipelago of 297 islands (≥ 5ha) inhabited by a native flora and fauna that evolved in the absence of terrestrial mammals (Atkinson and Cameron 1993). Mice first arrived in New Zealand in 1824 following a shipwreck and are now present across the whole country (Ruscoe and Murphy 2005) after multiple colonisation events from diverse sources (Searle *et al.* 2009). Because mice in New Zealand islands have detrimental impacts on native flora and fauna (e.g., Newman 1994; Miller and Miller 1995; Miller and Webb 2001; Wilson *et al.* 2007b), there have been 28 eradication attempts (Howald *et al.* 2007; MacKay *et al.* 2007), 16 of which succeeded and 12 failed (MacKay *et al.* 2007).

Information about mouse populations on New Zealand islands is scarce in the literature. There are few estimates of mouse population densities (White and King 2006) on 'mainland' New Zealand or on its offshore islands, and home range sizes and nightly movement distances have rarely been studied. This paper describes the first detailed study of a population of house mice during an eradication on a small New Zealand island. We used trapping and radio-tracking to determine densities and movements throughout the year and also collected demographic information about the population for comparison with other studies. These data were then employed to design a successful mouse eradication using trapping and poisoning during the Austral winter, when mouse eradications are typically attempted.

METHODS

This study took place on Saddle (Te Haupa) Island in the Hauraki Gulf, New Zealand (36˚31'S, 174˚47'E; Fig. 1). The island is long and narrow (650 m by 50–150 m wide; C. 6 ha), has steep cliffs around the littoral area, and reaches 35 m above sea level. Norway rats were eradicated from the island by poisoning in 1989 (Howald *et al.* 2007) and mice were detected shortly afterwards (Tennyson and

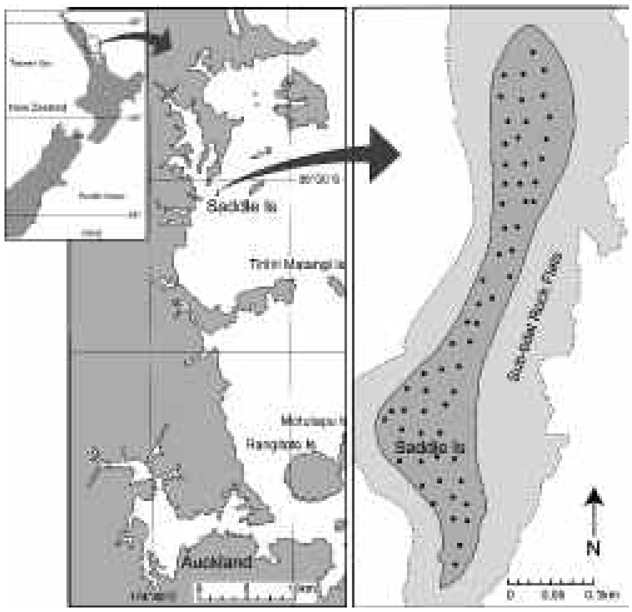

Fig. 1 Location of Saddle Island and locations of trapping stations.

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Table 1 Summary of trapping visits to Saddle Island, New Zealand. CMR=capture-mark-recapture

	Month	No. trap-nights	Purpose
1	January 2008	5	CMR
2	March 2008	4	CMR
3	May 2008	4	CMR
4	July 2008	1 (4 nights of telemetry)	Radio-tracking
5	August 2008	4	Removal trapping

Taylor 1999). It is not known whether mice were present concurrently with Norway rats or invaded following Norway rat eradication. Further details of the island's history, fauna and flora are provided by Tennyson and Taylor (1999).

We established a grid of 62 stations (Fig. 1) at 25 m intervals on the island in October 2007. This grid was used to place traps for live capture, stations for poison bait, other devices for monitoring mouse activity, and as an aid for navigation during night work. A Longworth live capture mouse trap (Chitty and Kempson 1949) was set at each station five times between January and August 2008 (Table 1). Each trap contained Dacron fibre for bedding, with peanut butter on a carrot disk and oats as bait.

Capture-Mark-Recapture protocol

Traps were checked daily during each four- or five-night Capture-Mark-Recapture (CMR) session. Captured mice were weighed, sexed, and had a numbered tag (National Band and Tag Co., Newport, Kentucky, USA) attached to each ear. After tagging, the animals were released at their capture site. The tag numbers of previously marked animals were recorded and the presence of torn ears was noted. Lost tags were replaced only when missing from both ears.

Radio-tracking

Traps were set to catch mice for fitting with radio collars on 16 July 2008 (Table 1) and captured animals were processed according to the protocol above. Only mice > 12 g were used for telemetry. At this weight the 0.6 g transmitters were ≤ 5% of mouse body weight and therefore unlikely to affect mouse behaviour (Pouliquen *et al.* 1990, Mikesic and Drickamer 1992). From the captured animals, four males and two females were selected for radio-tracking according to their capture location, to achieve a spread of animals across the whole island. Six animals were the maximum number that could be effectively tracked simultaneously. Animals were transferred to a plastic bag and anaesthetised with a piece of cotton wool soaked in isoflurane. As isoflurane is a rapid acting anaesthetic which wears off quickly, animals required two or three doses to fit the transmitter. Transmitters were a single stage whip aerial type (Model BD-2NC, Holohil Systems Ltd., Carp, Ontario, Canada), fitted by looping the aerial wire around the mouse's neck and crimping the wire to fasten it. Animals were returned to closed traps to recover. All animals, including those not selected for radio-tracking, were returned to their capture locations and released.

Radio-tracking began at 1800 h on 17 July 2008. Animals were tracked by two operators using TR4 receivers (Telonics, Mesa, Arizona, USA) with Yagi 3-stage folding antennas (Sirtrack Electronics, Havelock North, New Zealand). As most mouse activity was near the beach, tracking was most efficient when one operator walked along the beach while the other confirmed locations from the cliff-top above. When the animal was between the trackers its location was noted by recording a bearing and estimating the distance from a marked point on the beach. When an animal ventured into the interior of the island both people tracked the animal and a distance and bearing was recorded in a similar manner from the nearest trap site. Marked locations were then revisited in August and mapped with a GPS. Four or five fixes at approximately 90-120 minute intervals throughout the night and one daytime den site fix were obtained for each mouse over four nights of tracking. Some night fixes were missed due to adverse weather conditions. Daytime den fixes were confirmed by using the telemetry receiver without an antenna to maximise accuracy. To minimise disturbance, mice were not approached as closely at night as during the day. Despite this, the mice were often seen while being tracked, which confirmed the accuracy of night fixes.

Removal trapping using Longworth live traps was undertaken over four nights in August 2008 (Table 1). Captured animals were euthanased by cervical dislocation. Mice were then weighed, sexed, and any ear tags present from previous trapping sessions or ripped ears were recorded. A small piece of tail tip was taken from each animal and preserved in 70% ethanol for future genetic analysis. Such samples obtained before eradication attempts provide a means of distinguishing failed eradications from re-invasion should mice reappear (Abdelkrim *et al.* 2007; MacKay *et al.* 2007). A WaxTag (Thomas *et al.* 1999) baited with peanut butter was placed at each trap station on 7 August at the end of removal trapping and checked and removed on 19 August when poison was applied to the island. The locations of chewed tags, showing where mice remained following removal trapping, were recorded.

The anticoagulant toxin brodifacoum was applied to the island on 19 August 2008. Toxin was applied in two formulations: wax blocks (Pestoff Rodent Blocks) in bait stations, and approximately 15 kg of pellets (Pestoff 20R Pellets) spread around cliffs on the east coast, the north and south points and areas with dense shrub cover or mixed shrub, and open grassland on the west coast. Three wax blocks of toxin were wired to a tree under a plastic cover at each trap station to make improvised bait stations designed

Table 2 Monitoring visits to Saddle Island following poison application.

Date	Event
19/08/08	Poison bait distributed on the island in bait stations and hand-spread on cliffs.
16/09/08	Poison bait stations checked and location of chewed blocks recorded; WaxTags and ink tracking tunnels baited with chocolate nut spread deployed on alternate lines across island.
18/09/08	Detection devices checked; wax poison block placed in each tracking tunnel giving 31 more bait stations. Total bait density including pellets and blocks approximately 4 kg/ha
26/09/08	Poison bait stations removed from island; WaxTags and tracking tunnels left in place; poison in tracking tunnels left in place
03/12/08	Eradication confirmation with trained rodent dog Occi; poison removed from tracking tunnels; traps set around small area of possible mouse scent (since considered to be a response by the dog to skink scent (M. Ritchie pers. comm. 19/01/10))
15/12/08	Traps and devices checked

to shelter the poison blocks but to allow easy access to mice. Wax blocks in bait stations were not replaced and were removed from the island on 26 September 2009 (Table 2). Total bait density of wax blocks and pellets was approximately 4 kg/ha. Following poison application, the island was intensively checked (Table 2) using 31 ink-based footprint tracking tunnels (Gotcha Traps, Warkworth, New Zealand and Connovation Ltd., Auckland, New Zealand) and 31 WaxTags set at trap stations on alternate lines across the island. Two unsecured poison blocks were placed in each tracking tunnel on 18 September 2008 to create 31 further bait stations. These blocks were left in place until 3 December 2008 when the island was checked by a Department of Conservation rodent detection dog 'Occi' (handler: Miriam Ritchie). Rodent detection dogs are commonly used in New Zealand and around the world to aid in the confirmation of eradication success or failure (Gsell *et al.* 2010).

Analysis

Four estimates of mouse population size on the island were calculated using two methods. Estimates for January, March, and May were calculated using closed-capture models in program MARK (White and Burnham 1999). Trapping data from August were analysed using a removal trapping catch effort method augmented by independent index data from WaxTags to reduce bias (Russell *et al.* 2009). For this augmented removal estimate we assumed multiple mice could interfere with a single WaxTag. Analysis in MARK followed Wilson *et al.* (2007a), with three covariates used to model heterogeneity in the data. Two categorical variables (sex and age) and one continuous variable (weight) were used as covariates in four models incorporating both behavioural response to trapping and variation in capture probability between trap nights. Mice are difficult to classify as adults or juveniles based on external characteristics, so we classified animals weighing less than 12 g as juveniles. This weight was chosen based on the mean weight of non-fecund mice recorded during a study at nearby Tawharanui Open Sanctuary (Goldwater 2007). Six covariate combinations (none; sex; weight; age; sex and weight; sex and age) were tested for each model. The model-averaging procedure in MARK was used to calculate population estimates based on all models except those where parameters were identified as singular or standard errors of estimates were very large or zero. Confidence limits (95%) of the averaged estimate were adjusted to take into account the actual number of mice caught in each trapping session (White *et al.* 1999). Population estimates were converted into density estimates (mice/ha) by dividing the estimate by 6 ha, the area of the island. MARK was also used to obtain a rudimentary survival estimate. Capture data were pooled for all sessions (except the single night of trapping in July) to estimate monthly survival, maximum lifetime and mean lifetime.

Information on animal home ranges and ranging behaviour was collected through trapping records and radio-tracking. Home ranges were calculated for all individuals that were trapped five or more times, and trapping records for the radio-tracked individuals were combined with radio-tracking data to calculate home-range sizes for these animals. Average movements were described from radio-tracking data alone. Movement information was compared to habitat observations from the island to investigate whether different habitat affected movements. Home ranges were estimated using harmonic mean estimation in Ranges7 (South *et al.* 2005). We estimated a 95% range core to avoid outlying fixes biasing the range size estimate upwards (Moro and Morris 2000). Ranges7 was also used to summarise animal movements and to estimate the area of the island sampled by traps assuming each trap had a 'circle of influence' with a radius equivalent to the average male or average female between fix movements. The combined area of the circle of influence for each trap was compared with the total island area to obtain an estimate of the proportion of the island sampled by traps.

RESULTS

Demographics

Between January and August, 154 mice were caught and tagged on the island (Table 3). Many unmarked individuals entered the population in March resulting in a relatively low recapture rate which then generally increased through the year (Table 3). Many mice were captured only in a single session; six were caught in four trapping sessions, and none in all five. There was a relatively high rate of tag loss between trapping sessions and 41 mice lost both ear tags between trapping sessions. This meant that each session had to be treated separately in CMR analysis. Three mice caught in January were captured and killed in August, indicating that they were at least 8 months old at time of death. Six mice died in traps during trapping sessions prior to August and 51 mice were trapped and killed in August, leaving 97 animals of unknown fate. Assuming tag loss was random, rudimentary survival analysis gave a monthly survival estimate of 0.6, a maximum lifetime of 26 months and a mean life span of 5 months. Tag loss between sessions will have biased the survival estimate downwards.

Pregnant or lactating female mice (indicated by prominent nipples) were recorded only in January and March. By July, most animals caught were at least 12 g in weight and were classified as adults, which suggests that breeding had ceased at least a month earlier. The proportion of females caught tended to decrease through the year with females representing only 27% of the animals caught during removal trapping in August (Table 4).

Population size

Because models with age covariates consistently ranked higher than models with weight covariates (based on Akaike's information criterion; Burnham and Anderson 2002), weight models were deleted before model averaging. The estimated population size varied between 53 and 115 individuals and was highest in March (Fig. 2). Confidence intervals were wide for population estimates in January and March because of the relatively high number of animals caught only once in these sessions (42% and 52% respectively). In May, this group decreased to only 24%. The removal trapping and WaxTag dataset produced a population estimate with very narrow confidence intervals. This August population estimate was 53 animals, whereas 51 mice were actually removed. Mouse densities therefore varied between 8.8 and 19.2 mice/ha (Table 5).

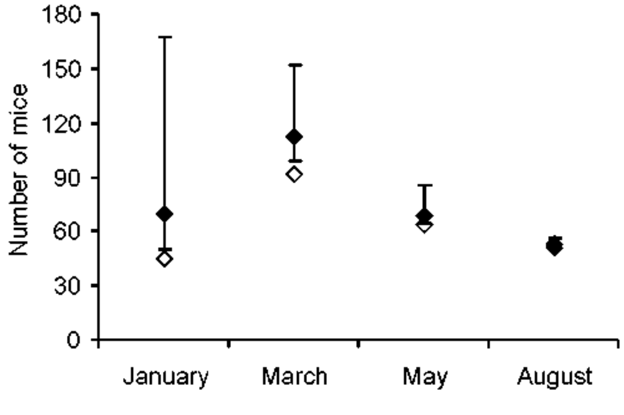

Fig. 2 Number of mice caught (open) and estimated population size (closed) with 95% confidence interval (CI) shown for each trapping session where population size was estimated.

Table 5 Mouse density for each trapping session on Saddle Island. *Density was not calculated in July

Month	Density (range)
January	12.8 mice/ha (8.5-36.2)
March	19.2 mice/ha(16.8-25.8)
May	11.3 mice/ha(10.7-14.3)
July	n/a*
August	8.8 mice/ha (8.7-9.3)

Ranging behaviour

Average home-range size (± SE) from radio-tracking data for female mice was 0.19 ± 0.04 ha (n=2) and for male mice was 0.38 ± 0.07 ha (n=4) (Table 6). Radio-tracked mice returned to the same den site at the end of each tracking night and males M2 and M3 had dens within 1 m of each other underneath the same karo (*Pittosporum crassifolium*) bush. Eighteen further home ranges were obtained using trapping information from animals that had been trapped five or more times (range 5–10 locations). Combining trapping and radio-tracking data gives average home range size (± SE) of 0.28 ± 0.05 ha for female mice (n=9) and 0.23 ± 0.03 ha for male mice (n=15). The animals with the smallest home ranges and lowest mean distance between fixes were in areas of the island with more understorey; generally with dense shrub cover or a combination of open grass and shrubs. Larger home ranges and movements were associated with more open areas of the island with sparse understorey.

Average movement between fixes (± SE) for radio-tracked females was 15.8 ± 7.0 m and for males was 24.9 ± 4.9 m. Five of the six tracked mice moved over 25 m at least once during the four-night tracking period, so were likely to have encountered a bait station (spaced 25 m apart). The sixth animal had a maximum movement between consecutive fixes of 23.5 m (Table 6). The maximum distance recorded between fixes was 142 m travelled by a male mouse in just over 2 h. Based on these values, GIS analysis suggested that the trapping grid 'circle of influence' covered 78.7% of the island for females and 95.7% for males.

Eradication and monitoring

Removal trapping ended on 7 August 2008. Of 62 WaxTags, 18 were chewed over 13 nights between the end of trapping and poison being laid on 19 August. Chewed tags were located between lines 1 and 7 at the north of the island and 15 and 23 at the south, with no sign of mouse activity in between. Poison bait take from stations was minimal; although 13 out of 62 bait stations showed signs of interference when they were checked on 16 September, only two of these showed conclusive signs of interference by mice and the remaining 11 could have been due to invertebrates. The distribution of bait take from bait stations closely matched that of chewed WaxTags. No further signs of mice were found after this and the eradication was confirmed as successful on 3 December (Miriam Ritchie, Department of Conservation pers. comm.). Ongoing monitoring throughout 2009 did not detect any mice other than those released deliberately during a study into mouse invasion behaviour (J. MacKay, unpublished data).

Rat incursions

In March 2008, rat sign was detected on the island and four DOC 200 traps were deployed. A large male Norway rat was captured on 14 May 2008. No further rat sign was detected until rat-tracked tracking cards were found on 3 December 2008. However, a trained rodent dog showed no reaction to the cards suggesting that the prints were older than 15 days, this being the length of time for which rodent scent persists (Gsell *et al.* 2010). No further evidence of rats has been found on the island. During mouse trapping in March 2008 (four nights) and May (two nights) an average of five mouse traps per night were pulled apart by the rat and were therefore unavailable for mouse trapping. All traps that had been pulled apart had mouse droppings inside them, so it appears that the rat was targeting traps that had caught mice.

DISCUSSION

Mice were successfully eradicated from Saddle Island, New Zealand, using a combination of removal trapping and poisoning. By gathering a large amount of data about the mouse population prior to eradication, we can now assess why the eradication was successful.

Demographics

The main predators of mice in New Zealand are stoats (*Mustela erminea*) and cats (*Felis catus*) (Ruscoe and Murphy 2005) both of which are absent from Saddle Island. Mouse population dynamics on the island were therefore influenced largely by food availability and climatic factors. Live trapping revealed a biased sex ratio of mice; 65% were males. During removal trapping in August, 73% were males. Male biased sex ratios have been recorded in some other trapping studies of mice in New Zealand (Ruscoe and Murphy 2005). The alternative scenario, that sex ratios of mouse populations are generally at parity but that trappability differs between the sexes (Efford *et al.* 1988), is not supported by our data. Our removal estimate of 53 mice on Saddle Island at the time of eradication, when 51 mice (37 males) were captured and removed, also supports the conclusion that in August there was a male bias in the population. The bias may have been caused by differentially greater mortality of females due to the physiological demands of breeding (Calow 1979).

Rodent eradication attempts generally occur in winter when natural food availability is low and rodent populations have declined (Howald *et al.* 2007). Mice do not normally breed over winter in New Zealand, except in mast seeding years (years where certain tree species produce vast quantities of seeds, Ruscoe and Murphy 2005). There was no evidence of mice breeding on Saddle Island over the study winter so it is unlikely that young animals were in nests and not exposed to poison bait.

Table 6 Summary of movement data obtained in July 2008 for six radio-tracked mice (M: males, F: females)

Animal	Number of fixes	First and last capture	Range area (ha)	Mean (±SE) distance (m) between fixes	Maximum distance (m) between consecutive fixes
F1	26	17/07-04/08	0.23	22.8 ±3.9	53.0
M2	24	17/07-21/07	0.43	32.8 ±7.8	142.0
M3	29	08/03-04/08	0.41	26.5 ±6.1	190.6
M4	29	06/03-04/08	0.48	29.6 ±5.3	72.3
M5	28	15/05-04/08	0.18	10.8 ±2.9	50.2
F6	24	08/01-04/08	0.15	8.8 ±1.7	23.5

Ranges, movements, and habitat

The average home ranges of animals recorded in this study fall in the middle range of those reported elsewhere in New Zealand. For example, in forest with multiple pests in the Orongorongo Valley, east of Wellington, mouse home ranges averaged 0.6 ha (Fitzgerald *et al.* 1981). At Tawharanui Open Sanctuary north of Auckland, where mice are the only rodent species present, home range lengths were <40 m (Goldwater 2007). One criterion for successful eradication is that every animal must be able to come into contact with a kill device (poison bait or trap) during their nightly movements (Bomford and O'Brien 1995). Although one female and one male mouse radio-tracked on the island had small core home ranges (0.15 ha and 0.18 ha respectively) and short mean (±SE) distances between fixes (8.8 ±1.7 m and 10.8 ±2.9 m respectively), they both also had larger movements outside their core home range (Table 6) and would therefore have been likely to come into contact with the poison grid. However, the effect of habitat on animal movements was quite striking; the average movement between fixes for two individuals from areas with denser ground cover and more understorey was half that of those from more open areas of the island. A similar effect was noted on the Isle of May in Scotland where mice living in open, 'featureless' areas had larger home ranges than those living in varied habitats with more cover available (Triggs 1991). When mice and ship rats were both present on Browns Island in New Zealand mice were only caught in areas of dense ground cover (Weihong *et al.* 1999).

Density

Estimates of mouse population density are rare in the literature (White and King 2006) and most studies report indices of mouse abundance rather than density (Ruscoe and Murphy 2005, *cf.* Wilson and Lee 2010). In the course of this study we calculated mouse density using three sessions of CMR and also with an index augmented removal estimate. Precise removal estimates are notoriously difficult to obtain (Russell *et al.* 2009), but our combination of trapping data and data from detection devices allowed mouse density to be calculated. Mouse density estimates on Saddle Island ranged from 19.2 mice/ha in March to 8.8 mice/ha in August. A similar seasonal pattern of density fluctuation through the year has also been observed on two other New Zealand islands where mice were the only introduced rodent species present. Mouse density estimates on forested Allports Island in the Marlborough Sounds ranged from 17 mice/ha to 2.2 mice/ha in September (Murphy 1989). Similarly, the mouse population on Mana Island near Wellington reached a density of 71 mice/ha in March and fell to 5.2 mice/ha in September (Pickard 1984). The highest mouse densities on Mana Island were found in grassy habitats, which may support higher densities compared with forests (Efford *et al*. 1988). Comparing populations on different islands in different regions is difficult as local climatic factors may also influence mouse population size.

The distribution of chewed WaxTags and poison bait take from bait stations was similar suggesting that mice remaining on the island following trapping were both detectable by WaxTags and susceptible to poison. The low incidence of poison bait uptake by mice at bait stations after the removal trapping session confirms that the population was small and most animals were removed through trapping.

CONCLUSIONS AND RECOMMENDATIONS

MacKay *et al.* (2007) suggested that mouse eradication failures may be caused by aspects of mouse behaviour. Here our eradication method of trapping was followed by poisoning and yielded a successful eradication. We also collected information about the population and individual behaviour of mice prior to eradication which allowed us to address why and how the eradication succeeded.

Habitat has a large effect on mouse home range size and their movement behaviour. MacKay *et al.* (2007) suggested that mice in complex habitats may have small home ranges and here we present data confirming this prediction. In areas where ground cover was dense, average movements between fixes and home range size were lower. Because of logistical constraints, sample sizes of tracked animals were low (n=6), but the resulting information was consistent with the live-trapping data. As part of eradication planning, areas of complex habitat should be identified and eradication methods adapted to ensure all mice living in these areas have access to bait.

We endorse the value of genetic samples collected before an eradication attempt to distinguish between failed eradications and reinvasions (Abdelkrim *et al.* 2007, MacKay *et al.* 2007). Combining removal trapping and detection devices allowed an accurate density estimate to be calculated (Russell et al. 2009) and if time and resources are available, a grid of snap traps could provide genetic samples and data to accurately estimate mouse population size.

Trapping followed by poisoning proved to be an effective method of mouse eradication on a 6 ha island. A 25 m grid was adequate in this instance, and five out of the six mice radio-tracked moved over 25 m between fixes at least once during a four-night tracking period. A 25 m grid of bait stations has been used to eradicate mice from 253 ha Flat Island in Mauritius (Bell 2002), but successfully scaling up to larger islands will depend on terrain and vegetation, as generating and maintaining a grid of traps and/or bait stations is very labour-intensive.

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Campbell Island – pushing the boundaries of rat eradications

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Abstract. Campbell Island (11,300 ha) is situated 700km south of New Zealand, making the project to eradicate Norway rats (*Rattus norvegicus*) the largest and one of the most isolated rodent eradications ever undertaken. The methodology for the project built on the techniques developed on a range of smaller island eradications closer to New Zealand. The size and isolation of Campbell Island meant that aspects such as the baiting rate and bait storage had to be significantly modified to make it financially and logistically feasible. The changes from the standard techniques were significant enough to require an extensive field trial in 1999 to test the probability of success. In 2001, the New Zealand Department of Conservation (DOC) used four helicopters to transport and spread 120 tonnes of bait containing 20ppm brodifacoum. While three months had been allowed to complete the project, the bait drop took just over four weeks due to unexpectedly good weather. The initial monitoring, using trained dogs, trapping and gnaw sticks was undertaken in 2003 and found no sign of rats. Continued opportunistic checks, along with outcome monitoring, has shown that the eradication has been successful. Several species of land and seabirds have recolonised Campbell Island since the eradication and some invertebrate populations have increased dramatically. This project has proved that increasingly larger and more isolated islands could be successfully cleared of rats providing there is sufficient funding; the required political and institutional support; a high level of planning to customise the methodology to the particular island; and a positive attitude from all those involved. It has already lead the way for other ground-breaking projects including: Rat Island, Aleutians (2008); Macquarie Island (planned for 2010); and South Georgia (planned to start in 2011), with hopefully more to come.

Keywords: Eradication, Norway rat, *Rattus norvegicus*, aerial

INTRODUCTION

Campbell Island (11,300ha) is located 700km south of New Zealand in the latitudes known as the furious fifties due to their consistently strong winds (Fig. 1). During the 19th century, the island was primarily a base for sealing and land based whaling. The island was also farmed from 1895 until 1931, after which the farm and its livestock were abandoned. The legacy of these activities included feral sheep (*Ovis aries*), feral cattle(*Bos taurus*), feral cats (*Felis catus*) and Norway rats (*Rattus norvegicus*). The island was subsequently designated as a Nature Reserve and is now administered by the Department of Conservation (DOC). Restoration of the island started in 1970 when sheep were removed from approximately half the island. Over the following 20 years, the remaining sheep and a small herd of feral cattle were removed (Brown 2002), leaving feral cats and Norway rats as the only introduced mammals.

This paper describes the methods used for what is still the largest successful rat eradication ever undertaken. I describe how existing methods for aerial spread of rat baits were tested and adapted for the island's remote location and difficult terrain, and outline the early responses to rat removal by native species.

Most people considered that it was not feasible to eradicate rats from Campbell Island because of its distance from the mainland, size and topography. Furthermore, since not all parts of the island could be safely accessed, a ground-based operation was impossible. The only solution was aerial spread of baits using helicopters. When the Campbell Island project was proposed, the largest previous aerial operation had been on 1,965ha Kapiti Island 5 km off the west coast of the North Island of New Zealand. Because baits could be ferried to Kapiti from the mainland, there were few logistic difficulties (Empson 1996; Empson and Miskelly 1999).

Initially, the Campbell project was for a joint rat and cat eradication (DOC 1998). However, given several years without any sign of cats, checks were carried out in 1999 using trained dogs. These confirmed that there were no cats present (Brown and Theobald 1999). It is not known why the cats died out, although it may be related to the previous removal of sheep. This was followed by increasingly dense vegetation cover (Meurk 1982) which either created difficulties when cats were hunting rats or, when wet, the vegetation was too inhospitable for cats. The absence of cats greatly simplified the eradication project.

METHODS

Bait trials

In the late 1990s, the accepted method for aerial spread of baits against rats in New Zealand involved two bait drops of 8 and 4 kg/ha with a ten-day gap in between and a forecast of three fine nights following each bait drop. Initial planning for Campbell Island indicated that the amount of bait required for this conventional approach, and the costs of transporting and spreading it, were not affordable. Similarly, the number of hours of suitable weather required for spreading a total of 12 kg/ha of bait was unlikely within the time available. The baiting rate was thus reduced to a single drop of 6 kg/ha involving 3 kg/ha out of the bait spreader with a 50% overlap of bait swaths to minimise the risk of gaps. Although based partially on previous experience, the chosen rate was largely based

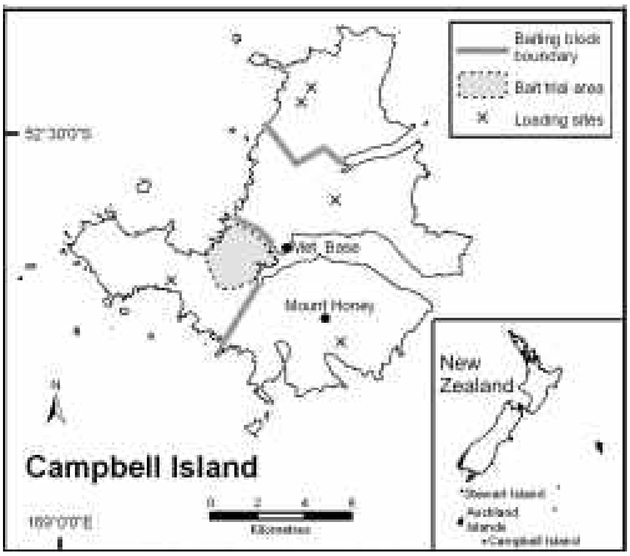

Fig. 1 Campbell Island.

Pages 204-207 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

on the gut feelings of the advisory group and as such it needed testing for its efficacy on Campbell Island. In 1999, a trial was carried out with non-toxic 16mm Pestoff cereal pellet baits containing the biomarker Rhodomine spread at the proposed rate over approximately 600 ha containing most of the habitat types found on Campbell Island (Fig. 1, McClelland *et al*. 1999). Snap-traps were then set throughout the area. All of the rats caught, except two near the boundary of the test block, had eaten the bait. Since the latter two rats were likely to have moved into the area after resident rats had been removed during the trapping programme, bait uptake was considered sufficient to indicate the potential for 100% mortality.

Compared with islands elsewhere around New Zealand where rat eradications have been undertaken, the climate on Campbell is wetter and has a high probability of snow, at least for short periods. The type and size of baits thus needed testing for the conditions on Campbell Island (Brown 1999). "Pestoff 20R" 16 mm cereal pellets performed best in the cool wet climate. Brodifacoum, a second generation anticoagulant, was chosen as the preferred toxin as it can kill rats after a single feed. All previous rodent eradications using helicopters around New Zealand had used baits containing this toxin with minimal failures.

Non target species

Experience from previous bait drops against rats, and the feeding behaviour of potential non-target species, indicated that albatrosses, penguins, seals, and sea lions were at minimal or no risk of either primary or secondary poisoning. As no native land birds remained on the island, the only species believed to be at risk were Subantarctic skua (*Catharacta antarctica*), southern black-backed (*Larus dominicanus*) and red-billed (*L. novaehollandiae*) gulls, self-introduced mallard ducks (*Anas platyrhynchos*), a variety of self introduced passerines, and possibly northern giant petrel (*Macronectes halli*). Although giant petrels were of concern because their populations are in global decline, as scavengers they were at low risk of either primary or secondary poisoning. Furthermore, no feasible precautions could be taken to protect them.

Timing

Timing of the operation was based on the standard assumptions used for most previous New Zealand rat eradications: that bait should be spread in winter when 1) natural food sources were minimal; 2) rat numbers are lowest thus requiring less bait to give all rats access to a lethal dose; and 3) there is least chance of rats breeding with the associated risk of young rats emerging from their nest after all bait had been consumed or decomposed.

Colonies of grey-headed (*Diomedea chrysostoma*) and Campbell (*Diomedea impavida*) albatross and rockhopper penguins (*Eudyptes chrysocome*) on the island were potentially at risk of disturbance by the helicopters, and the albatrosses could pose a risk of air strike. It was decided that the project should be carried out during the winter absence of these species from the island. Furthermore, a winter operation would remove the risk of primary and secondary poisoning of skua, which are also absent from the island at this time.

The combination of maximum effects on rats and minimum risk to other species narrowed the period to three months in which to complete the bait drop.

Logistics

Once the baiting regimes were decided, and given that there were minimal non-target issues, the main planning tasks involved the following numerous inter-locking logistical issues.

Helicopters. Three helicopters were used to spread the bait in order to maximise the chances of completing the drop within the three month period available, ie to allow for anticipated poor weather, and to provide cover in the event of a breakdown of a machine or illness of a pilot. Bell Jet Ranger 206s were selected because: 1) they were the most readily available model of helicopter already equipped for bait sowing, 2) most of the experienced bait sowing pilots were familiar with these machines, and 3) they were proven for their reliability and relatively easy maintenance. The last two criteria were especially important given the conditions expected on Campbell and the limited maintenance facilities available.

A spare bait spreading bucket and GPS base station as well as a range of spare parts for the helicopters and buckets were taken to the island, but were not required.

The bait pods (see below) weighed 850kg, which is beyond the lifting capacity of a Jet Ranger, so an Aerospatiale B2 Squirrel was used to move them and transport personnel around the island. A second Squirrel was taken to the island to ensure that the ship was unloaded within one day to reduce ship charter costs. The second Squirrel then returned to New Zealand. The extra cost of two Squirrels paid for itself because after unloading was completed as planned, the following three days were unsuitable for flying.

Pilots. The selection of pilots is a crucial part of any aerial eradication, especially when operating in a harsh and remote environment like Campbell. The lead pilot (Peter Garden) was selected for his constructive attitude, experience with three previous eradication bait drops and numerous control operations on the mainland, and expertise in the use and downloading of the GIS information. Peter assisted with planning and in turn selected the other two Jet Ranger pilots (Brian Beck and Don Sanders) based on their experience, skills and attitude. The squirrel pilot Richard (Hannibal) Hayes was recognised as one of the best in New Zealand for the long-line work required to unload the ship and move the pods to the loading sites. He was also very experienced at mountain and poor weather flying. The four pilots all worked for different helicopter companies. Because agricultural helicopters are often set up by each company differently for doing the same job, each company provided the specific helicopter with which the pilot was most familiar. This reduced the risk of unforeseen problems.

Infrastructure. The New Zealand Meteorological Service built a weather station and associated accommodation on Campbell Island in the 1950s. The station was automated and destaffed in 1995 but the buildings were still in good condition, although stripped of chattels. An advance team set up the base, including electricity, heating, hot water, and gas cooking prior to the arrival of the main party, helicopters, and the bait. A warm, dry environment to return to each day meant that the team was comfortable and able to work harder longer and in worse conditions than might otherwise have been the case. It also meant that bait spreading teams would be ready to make the most of any suitable weather from the day they arrived.

Transport. Options for shipping the bait to the island were limited. The vessel used was the coastal freighter, "Jenka" with large holds and wide hatch covers to facilitate unloading of the bait by helicopter. The bait pods and other materials were stored in the hold of the vessel while the helicopter fuel was stored on deck to reduce the risk of bait contamination.

Shipping the helicopters to Campbell would have required a larger and more expensive ship and the pilots decided that it was safer to fly the machines to the island, thereby avoiding the need to remove blades or the risk of damage in transit. The helicopters were all fitted with long range fuel tanks which allowed them to make the flight utilising two fuel depots which had been set up the previous summer on islands on the way.

Bait transport and storage

None of the existing buildings on the island were suitable for storing the required 120 tonnes of bait. The storage needed to be waterproof, strong enough to hold more than half a tonne of bait, moveable by forklift, and able to be slung under a helicopter. The chosen option was a purpose-built 1.2 m^3 plywood box, termed a "pod" to avoid confusion with other containers and boxes used for the project. The pods weighed only 100 kg and could hold 750 kg of bait (25 x 30 kg bags); were easy to fly under a helicopter; enabled minimal bait handling because bait in filled pods wasn't touched until it was loaded into the spreader bucket; and they were comparatively cheap.

The pods had the added advantage that four could be formed into a loading platform of ideal height for loading the bait buckets. If bait spreading was stopped for any reason the bags of bait were returned to a pod, which was then resealed. When empty, the pods could be rapidly dismantled and flown back to base.

Baiting the blocks. Campbell Island is roughly triangular in shape so the island was divided into four blocks based around major geographical features: the three main ridges and the highest hill on the island (Fig. 1). This allowed for the most efficient management of the loading sites. Baits were sown within the blocks sequentially from the north so that areas around albatross colonies were covered before the birds returned to breed.

Monitoring the spread of baits. The spreading of baits was guided by differential GPS with a base station set on a high point to receive satellite signals. Differential GPS was found to reduce the risk of inconsistencies with the flight lines. The GPS units allowed each block to be divided up into numbered 40 m parallel swaths. These were then allocated among the three helicopters to ensure no swaths were missed or flown twice. Every evening after bait had been spread flight lines from the three helicopters were downloaded, printed and checked for gaps. Actual or potential gaps became priority work for the next day.

Loading sites. Multiple loading sites were established in order to minimise ferrying time for the bait-spreading helicopters and maximise the time they spent spreading baits. Six loading sites, five remote and one at the base, were used during the project. With loading teams working at two of them at any one time. Where the sites had soft peat soils, a working base of timber and dismantled pods was laid out prior to putting the four pods in place to make the loading platform.

When operating at the north end of the island, a second refuelling site was also set up. All other refuelling was done at the base.

Bait pods and fuel were ferried by the Squirrel helicopter from storage at the base to the loading sites being used at the time. Dismantled pods were stacked in cargo nets and back loaded to the base where they were packed for return to New Zealand.

Safety

Safety was a major concern for this project because of isolation, extreme weather, and the presence of four helicopters working simultaneously over the island. While helicopters were the greatest hazard, they also provided some reassurance that rapid evacuation of anyone injured was possible. Safety was stressed at every briefing and while there was an assigned safety officer, everybody was made responsible for both individual and team safety.

Field team selection

The project manager had full control over selecting the field team and ensuring compatibility within the group. A list of the required skills and experience was drawn up and the best people then targeted for those roles. The skills required included mechanical, electrical, cooking, medical, as well as experience with eradications and the Campbell Island environment.

Island Eradication Advisory Group

DOC's Island Eradication Advisory Group (IEAG), set up to advise multi-island eradication programmes (K. Broome pers. comm.), was involved in all planning phases for the Campbell Island operation (Broome *et al*. 1999). The IEAG ensured that the lessons learnt from previous eradications around New Zealand were considered during the planning for Campbell and that the lessons learnt from Campbell have been considered in subsequent projects.

RESULTS

Bait coverage

There were no gaps in the bait coverage due to the combined benefits of compatible GPS systems in all helicopters, careful checks of all flight lines after each day's bait spread, and the 50% overlap of flight lines. After the baits had been spread, a second check of the flight lines revealed several relatively small (50m x 200m) areas where bait had been applied at rates lower than expected. Apparently the 50% overlap had not been complete leaving bait at only 3 kg/ha in some patches. Ground checks showed that bait cover was still sufficient so no further action was undertaken.

Areas with cliffs, some of which were over 400m high, were baited with swaths flown parallel to the cliffs at intervals of approximately 100 vertical metres. The extent of coverage could not be confirmed in these areas so the helicopter pilots determined visually whether sufficient bait was landing on the ledges. A sideways deflector, which is a shield allowing bait to go out on only one side of the bucket, was trialled but not used due to mechanical problems.

Non target species

Searches for non-target mortality after the bait drop was opportunistic while doing other work. The only confirmed casualties were one mallard duck, 10 red-billed and black-backed gulls and 10 introduced passerines, most of which were redpolls (*Carduelis flammea*). There were no recorded effects on giant petrels.

Results monitoring

The first monitoring of the effects of baits on the rats was carried out over the 2003/2004 austral summer using snap traps, gnaw sticks and a trained rodent detection dog (King 2003). No sign of rats was detected. While too early to be sure of success, this gave sufficient confidence for the reintroduction of Campbell Island teal (*Anas nesiotis*) in 2004 (Gummer 2004). Additional low level monitoring using gnaw sticks and searching for sign in 2004 and 2005 failed to reveal any sign of rats. On the basis of these results, the eradication was declared successful in 2006. While the standard period before declaring success for an eradication is two years, this was extended for Campbell due to the size of the island and relative low intensity of the monitoring.

Outcome monitoring

Since the eradication of rats, the abundance of a flightless invertebrate, the weta (*Notoplectron campbellensis*), has increased dramatically (pers. obs.). In addition, Campbell Island pipits (*Anthus novaeseelandiae*) and Campbell Island snipe (*Coenocorypha aucklandica*) have recolonised Campbell from smaller rat-free islets offshore (Barker *et al*. 2005, Thompson *et al*. 2005). Grey-backed storm petrels (*Oceanites nereis*) (T Shaw pers. comm.) and white-chinned petrels (*Procellaria aequinoctialis*) (M Rutherford pers. comm.) have also both been recorded as breeding on the island for the first time.

DISCUSSION

There has been some suggestion that success of the Campbell Island project could be put down to the lucky break of "relatively" good weather. This enabled completion of the spread of baits in one month rather than the three months anticipated from previous weather records. Another view is that: Luck is when opportunity meets good planning. The Campbell Island project succeeded within the compressed time frame because of attention to detail when planning and the willingness and ability of the team, especially the pilots, to make the most of suitable weather. A conventional approach to spreading baits would have used a forecast of three nights with no precipitation and relatively calm weather. Under this regime, it is unlikely that the project would have been completed. Instead, every opportunity to spread bait was taken; any suitable weather window of two hours or more was regarded as sufficient to begin operations. This rapid response to local conditions also kept the baiting front progressing, which reduced the risk of rats reinvading areas where baits had already been consumed. Above all else, the successful aerial spread of baits reflected the skills and experience of the pilots and the precision with which they used GPS.

The Campbell Island project required a rethink of accepted aerial eradication methodology. Subsequent to the Campbell project, DOC has retained the well tested method of spreading bait at 8 kg and then 4 kg/ha. This approach avoids the potential risk of failure from reducing the baiting rate for rat populations that are likely to be at relatively high density. The Campbell eradication built on many years of knowledge developed over an extensive eradication programme around New Zealand. Other countries that have multiple islands, should look at treating their eradications as a programme to develop their techniques rather than simply tackling them one by one.

Removing rats from Campbell Island was a major achievement. It was built on many years of knowledge developed over an extensive eradication programme around New Zealand where each project was seen as an opportunity to refine techniques. As a result, the Campbell Island project demonstrated that it is possible to eradicate rats from increasingly larger and more isolated islands. Prerequisites for success were political and institutional support, adequate funding and a positive attitude. This success has not been lost on the international community. The eradication of rats from Campbell has already stimulated the eradication of rats from Rat Island over 2000km along the Aleutian chain (Bucklew *et al*. 2011). Planning is now underway for even larger islands such as Macquarie (Springer 2011), South Georgia and Gough Islands (Poncet *et al*. 2011).

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Trophic considerations in eradicating multiple pests

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Abstract Invasive species can fundamentally alter island ecosystems, and eradication is often necessary to abate the threats they pose to native species. But just as the introduction of a species to an island can have profound ecological effect so too can its removal. While significant ecological effects are often the desired outcome of an eradication, unintended or undesired effects can also manifest. One such undesired effect may be that due to the ecological change following removal of a pest, a different pest becomes more of a threat or more difficult to manage. Such risks may be reduced by eradicating multiple pests simultaneously, or if sequentially, in a manner that anticipates trophic cascades and first exploits the ecological impact of one pest to help render another more susceptible to control. To illustrate, I present a case study from Santa Cruz Island (250 km^2), 40 km off Santa Barbara, California, USA. For nearly two centuries, non-native species caused widespread destruction of natural communities, until recent decades when the most damaging of them were removed – some sequentially, others concurrently. I review that history and outline strategic considerations based on the ecological relationships of the managed non-native species, which include sheep, pigs, golden eagles, and wild turkey. This case study highlights how addressing pest management issues comprehensively can not only reduce cost and investment risk in island restoration programmes; it can also sooner abate key threats to often unique and often imperilled island biota.

Keywords: California, efficiency, eradication, planning, risk, Santa Cruz Island

INTRODUCTION

Invasive alien species can devastate island ecosystems and eradication is often necessary to protect native biota. Species targeted for eradication are typically those that have a profound impact on island resources, so it follows that their removal may have similarly profound effects. Dramatic ecological responses following eradication programmes have been observed (e.g., Howald *et al*. 2010), ranging from desired to undesired. These responses may or may not include those anticipated when the eradication was planned.

Perverse outcomes of eradications are perhaps more likely when there are multiple invasive species and the removal of one favours another (Zavaleta *et al*. 2001). Given the pervasiveness of invasive species and the severity of their impacts on naïve ecosystems, many, if not most, islands face multiple challenges from invasive species in need of management. In these situations, managers must determine how to invest limited funds to maximise desired outcomes, while minimising those that are undesired and or unexpected. However, this planning is frustrated by imperfect understanding of the myriad direct and indirect interactions in ecological communities. Modelling can be informative (e.g., Russell *et al*. 2009), but also unlikely to capture the full array of synergistic relationships, trophic complexity, and management constraints. Case studies can also be illuminating, although compared with single species efforts, the literature contains few examples where multiple invasive species have been managed.

Here I provide an overview of efforts to manage multiple invasive species over three decades on Santa Cruz Island, California, USA. Some of these pests were managed in series, others more or less simultaneously. Reviewing that history provides an opportunity to examine how understanding and exploiting the trophic relationships among pests and native flora and fauna can reduce the risks of perverse outcomes in eradication, and increase the efficiency of pest management and therefore island restoration.

CASE STUDY: SANTA CRUZ ISLAND

Santa Cruz Island is the largest of the eight Channel Islands off mainland southern California. The 250 km^2 island is co-owned and managed by The Nature Conservancy (TNC) and the United States National Park Service (NPS). The island has two rugged mountain ranges flanking a fault valley and experiences a Mediterranean-type climate of cool, wet winters and warm, dry summers. Vegetation communities are predominantly grassland, coastal scrub, chaparral, oak woodland, and pine forest. Four terrestrial nonvolant mammals are native to the island: island fox (*Urocyon littoralis santacruzae*), island spotted skunk (*Spilogale gracilis amphiala*), deer mouse (*Peromyscus maniculatus santacruzae*), and western harvest mouse (*Reithrodontomys megalotis santacruzae*).

For much of the past two centuries, Santa Cruz Island was used for ranching and agriculture. Sheep (*Ovis aries*) and pigs (*Sus scrofa*) introduced in the 1850s soon established feral populations that ranged throughout the island. Seven wild turkeys (*Meleagris gallopavo*) were introduced to the island in 1975. In 1978, TNC acquired 90% of the island. Channel Islands National Park was established in 1980, and in 1997 NPS acquired the remaining 10% of the island. Pest problems facing managers ranged from feral honeybees (*Apis mellifera*) and cattle (*Bos taurus*), to noxious forbs and weedy trees. Remarkably, there are no non-native rodents or feral cats.

Direct and indirect impacts of non-native ungulates have been implicated in threats to the survival of at least nine species of plants on the island (NPS 2002). In the late 1990s-early 2000s, the island fox population also underwent a precipitous decline. Golden eagles (*Aquila chrysaetos*), which had not previously been resident on the island, established a small population, likely due to an abundant food supply provided by feral pigs (Roemer *et al*. 2002). Incidental predation by eagles led to the Santa Cruz Island fox being listed as federally endangered in 2004.

Over the past 30 years of conservation management of the island, numerous programmes have been implemented to remove pests. Below I discuss some of those efforts and highlight lessons that may apply generally to island managers facing a similar need to control multiple species. The case study provides illustration of two general approaches to management: managing populations of invasive species in series, and managing them more or less simultaneously.

Managing pests in series

Sheep and cattle caused extensive degradation and destruction of native vegetation (Van Vuren 1981). In the 1980s, sheep were eradicated from 90% of the island (Schuyler 1993) and in the late 1990s from the remaining 10% of the island (Faulkner and Kessler 2011). Cattle were removed in 1988.

Release from herbivory triggered a dramatic vegetation response that had cascading ecological effect. Many native vegetation communities rebounded. For example, in 1985 bare ground and grassland covered nearly three-quarters of

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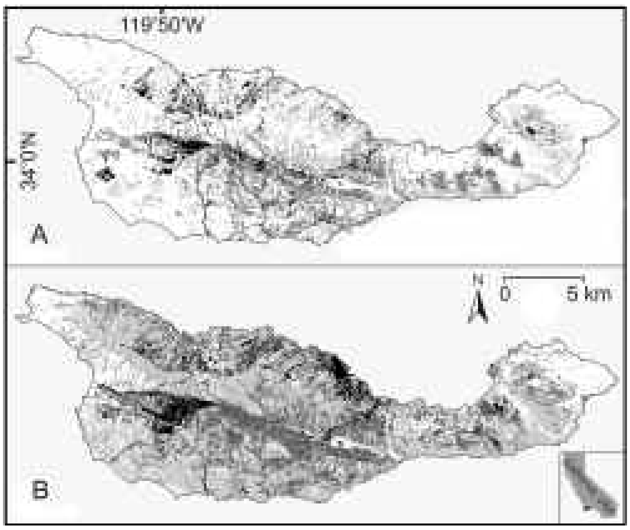

Fig. 1 Vegetation change on Santa Cruz Island, 1985-2005. Maps depict vegetation coverage, pooled into general categories: bare ground and herbaceous vegetation, white; scrub and low stature vegetation, light gray; chaparral and medium canopy communities, dark gray; forest and woodland, black. (A) Vegetation map prior to/during the eradication of feral sheep (adapted from Jones *et al*. 1993 and Howarth *et al*. 2005) (B) Vegetation map classified from a 2005 image (adapted from Cohen *et al*. 2009). Inset shows location of the island in the State of California, USA.

the island but by 2005 nearly three-quarters of the island was covered by native scrub, chaparral, and woodland vegetation (Fig. 1). This release from grazing pressures also likely contributed to a population explosion of invasive introduced fennel (*Foeniculum vulgare*) (Klinger *et al*. 1994). Feral pigs perhaps facilitated its spread via disturbance of soil and dispersal of seed (NPS 2002).

Until their removal from the island in 2005-2007, feral pigs likely benefited from increased vegetation cover that developed after the removal of sheep. This is speculative because there was no consistent monitoring of pigs before and after the sheep eradication. However, comparison of results from the pig eradication project on Santa Cruz Island with those from the neighbouring Santa Rosa Island may provide some clues. In the early 1990s, pigs were eradicated from 215 km^2 Santa Rosa Island (Lombardo and Faulkner 2000). At the time of that eradication, vegetation on the island was highly degraded, due to the grazing of introduced cattle, deer (*Odocoileus hemionus*), and elk (*Cervus elaphus*). The vegetation characteristics on Santa Rosa Island (i.e., >70% grassland or bare ground) resembled those of Santa Cruz Island in the 1980s before sheep were removed (Fig. 1). The two pig eradication efforts were roughly similar duration and on islands of roughly similar size, but they yielded only 1175 pigs on Santa Rosa compared with 5036 on Santa Cruz Island. Some of that difference likely owes to conditions being more droughty on Santa Rosa Island prior to the onset of that eradication effort; high inter-annual variability in rainfall and so productivity is characteristic of this semi-arid region, and pig populations can fluctuate greatly with resource availability (Beiber and Ruf 2005). But some of the difference in population size might reflect differences in habitat quality between the two islands at the time of respective efforts. Perhaps if pigs were removed from Santa Cruz Island either before or roughly contemporaneously with the sheep removal, the initial population of pigs might have been smaller – and fewer animals would have needed to be dispatched.

Even if the pig population was not smaller before vegetation recovery began on Santa Cruz Island, there would have been greater efficiency of eradication at that time because there was less vegetation to conceal the pigs. On Santa Cruz Island, 77% of the pigs were dispatched by a shooter from a helicopter (Parkes *et al*. 2010). Given that aerial hunting is more efficient in open habitat, having more open habitat (Fig. 1) would likely have led to a programme that was more cost and time efficient.

Meanwhile, throughout the 1980s and 1990s, the population of wild turkey remained at 40-50 birds in the vicinity of the initial introduction (P. Schuyler pers. comm.). In the early 2000s, however, their numbers increased >5-fold from 46 in 1999 to 276 in 2006 (L. Laughrin, UC Santa Barbara, unpublished data). Various trophic relationships may have contributed to that increase: turkeys may have been released from top-down control following the decline of the island fox population; perhaps turkeys benefited from a bottom-up increase in resources with the recovery of native vegetation following the sheep eradication, decades prior, and the island was turning into better turkey habitat (Fig. 1). Although its cause was unknown, the turkey population trend was especially problematic with pigs having just been removed from the system. Feral pigs are opportunistic omnivores (Wilcox and Van Vuren 2009) that likely competed with turkeys for food such as acorns, invertebrates, and small vertebrates. The pigs also probably depredated eggs and poults of turkeys. Without pigs and with habitat quality improving, turkeys had few limits on abundance and dispersal, and so were on a trajectory of becoming more difficult to manage. Managers were concerned that a large population of turkeys, also opportunistic omnivores, could directly affect island resources and also have potentially serious indirect impacts as another food subsidy for golden eagles, which would exacerbate the threat to foxes (Fig. 2). For that reason, an intensive control effort was launched in 2006 (Morrison 2007). Monitoring suggests that today only two male "sentinel" turkeys remain on the island (unpublished data).

Managing pests concurrently

Direct and indirect relationships among pests brought a convergence of crises to Santa Cruz Island in the early 2000s. Feral pigs were pushing a number of plants precariously close to extinction, and their presence was subsidising a population of golden eagles that was driving the island fox to a similar fate (NPS 2003). In 2003, the fox population was estimated to be less than 100 (NPS 2003).

Multiple strategies were used to manage these issues (NPS 2003). In 1999, live capture and removal of golden eagles was initiated. In total, 32 free-flying eagles were captured, mostly in the first years of the programme; detection and capture efficiency declined considerably as the population was reduced (SCPBRG 2004; IWS

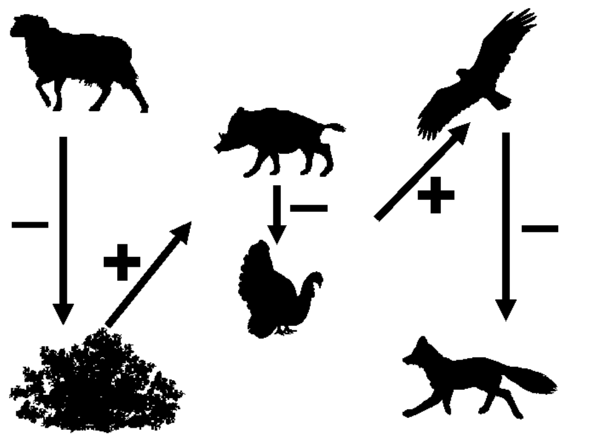

Fig. 2 Hypothesised trophic relationships of focal non-native and native species on Santa Cruz Island: sheep, native vegetation, feral pig, wild turkey, golden eagle and island fox.

2006). In 2002, 12 foxes were placed in fenced enclosures to protect them from predation and to launch a captive breeding program; eventually the captive breeding program expanded to house approximately 20 pairs of foxes. Also in 2002 a programme was initiated to re-establish bald eagles (*Haliaeetus leucocephalus*) on the island (Sharpe and Garcelon 2005). Bald eagles had been extirpated from the Channel Islands in the mid-1900s likely due to pesticide contamination from marine-sourced food. It was hypothesised that territorial behaviour of the mostly piscivorous bald eagles could deter golden eagles from settling on the island (NPS 2002). Indeed, agonistic behaviour between these species of eagles became a key component of the overall fox recovery strategy.

Long-term success, however, depended upon removing feral pigs from the system. In 1999, preparations began for an eradication programme (Morrison *et al*. 2011). Planning for the eradication included assessments of ways in which trophic relationships might affect the likelihood of attaining the various management goals. For example, there was concern that the pig eradication could impair golden eagle capture efforts, because an abundance of carcasses might make baiting even less effective than it had already become. To prevent carcasses of dispatched pigs from becoming a resource or attractant to eagles, the eradication contractor was required to move carcasses of any pigs dispatched in open areas into more densely vegetated areas where they would be concealed from foraging eagles. The greatest concern, however, was that the removal of the prey base provided by feral pigs would result in an intensification of hunting on island foxes by golden eagles. Some models suggested that that increased predation of foxes could have catastrophic consequences (Courchamp *et al*. 2003), leading some to advocate delay of the eradication program until all golden eagles had been removed from the island.

But delaying the eradication also had risks: What if the remaining eagles simply could not be captured (or killed, as some recommended)? What if removing pigs was in fact a prerequisite to being able to manage the last of the eagles? What if postponing the pig eradication effort jeopardised the ability to implement it at all, because the enabling factors that eradication projects need to succeed (see Morrison *et al*. 2011) would be difficult to reassemble? Managers assessed such questions on an ongoing basis. Ultimately, they decided to manage risks by advancing on multiple fronts: captive breeding of foxes; radio-collaring and frequent monitoring of a large portion of the wild fox population; population modelling and management planning for foxes (e.g., Bakker and Doak 2009); continuing efforts to capture golden eagles; removing the prey subsidy of eagles (feral pigs); re-establishing bald eagles; and maintaining materials on island to house more foxes in the event that predation rates became unsustainable and more foxes needed to be brought into temporary protective shelter. Thus, efforts to capture the remaining golden eagles were concurrent with the feral pig eradication.

As the onset of the pig eradication programme approached, sightings of golden eagles became exceedingly rare; their presence was mostly indicated by mortality signals from radio-collars of dead foxes. Spatial and temporal patterns of dead foxes were used to hone searches for eagles, estimate activity centres, and find nests. Nests were an important component of the capture strategy because the behaviour of nesting eagles was more predictable and so exploitable. In 2006, the nest of the last known breeding pair on the island was located, within days of egg hatching (IWS 2006). Although the removal of the young from the nest would have eliminated the eagles' immediate need to provision transportable (i.e., fox-sized) prey, nesting was allowed to continue to improve the likelihood of ultimately catching the parent birds. Monitoring of radio-collared foxes was intensive during this period, and it revealed a growing tally of dead foxes from the vicinity of the nest. However, population models indicated that the overall fox population could withstand that associated increased mortality. Approximately seven weeks later, both parent birds (and their chick) were captured and removed from the island (Morrison 2007). The remains of 13 island foxes were found in the nest (Collins *et al*. 2009).

The strategy of multiple concurrent pest management efforts appears to have been successful. Today, pigs are gone; there is no evidence golden eagles breed on the island; all of the foxes have been released from the captive facilities; and resident bald eagles breed on the island. The fox population is intensively monitored, and even though foxes are still occasionally depredated by (presumably transient) golden eagles, the fox population now shows very high annual survival rates (96.2% +/- 0.022, 80% CI) and is rebounding (736 +/- 254 adults, 80% CI in 2008; V. J. Bakker, unpublished data).

It is important to underscore that none of these efforts was guaranteed to succeed, and the fox-pig-eagle management crisis put considerable strain on the capacity of island managers. Facing such uncomfortably high stakes and dynamic circumstances, managers were fortunate to have a diverse group of external scientists and partners providing perspective (and often spirited discussion) on various alternatives for management. As managers were ultimately accountable for their decisions, having relevant and constructive input was invaluable for the necessarily adaptive implementation of the programme.

Current emphases in pest management

Santa Cruz Island is now free of un-managed introduced mainland vertebrates. Intensive monitoring of island foxes continues. This includes maintaining an array of radio-collared foxes that serve as "sentinels" for predators and disease (Bakker and Doak 2009). With the near decade-long extinction crisis at bay, the focus can now move to other resource management priorities, such as revising biosecurity protocols to protect investments. Now that soil disturbing pests (pigs) are out of the system, comprehensive weed management programmes are underway (Knapp *et al*. 2009) with greater confidence in enduring returns.

Vegetation recovery on the island (Fig. 1) should bring continuing benefits to native species. For example, increased tall vegetation will likely reduce the vulnerability of foxes to aerial predators. Nevertheless, there is a need to remain vigilant for undesired effects. Wildfire, for example, was probably historically uncommon on the islands (Anderson *et al*. 2009). However, increasing fuels, including flashy non-native grasses, and human activity (e.g., public access) may increase likelihood of ignition. Vegetation recovery may yet usher in other trophic cascades involving pests. For example, invasive Argentine ants (*Linepithema humile*) currently occur in a few localised infestations on the island (see Randall *et al*. 2011). Argentine ants are limited by availability of water and sugars, and actively tend honeydew producing aphids and scales. As shrub cover increases, so might vegetation suitable for honeydew producing species. Increased higher statured vegetation may also increase water inputs into the island ecosystem via moisture capture from fog – an otherwise desired positive feedback cycle for the island, but one that could also facilitate the invasion of Argentine ants.

HINDSIGHT

If we were to go back in time on Santa Cruz Island with the technological and methodological sophistication available in today's eradication "tool box", and ask how best to invest (always) limited resources to restore the island, we might ask two questions. The first is whether there is a "trophically strategic" sequence that the myriad pest issues should be engaged. That sequence would be aimed at reducing the potential for pests to contribute to perverse outcomes, and at using the impact of one species to render others easier to control. In this regard, sheep might have been considered a "keystone pest" on Santa Cruz Island as they suppressed weeds and probably affected habitat quality for pigs and turkeys. With hindsight, the turkeys should have been removed when they were still incipient invaders. If pigs were also removed ahead of sheep, it is possible that their numbers might have been lower and the feasibility of hunting and monitoring would have been enhanced due to the greatly reduced cover. While it is doubtful that fennel could have been fully eradicated, management to contain its spread was surely possible.

The second question we might ask is whether there would be benefits in engaging the pests concurrently. A comprehensive and concurrent approach would have had numerous benefits, including prevention of some of the observed perverse cascades, setting the island sooner on a recovery trajectory, and cost efficiency. Concerning the latter, when managers of Santa Cruz Island hired a professional wildlife management team to conduct the feral pig eradication, they brought to the island specialised personnel and equipment, including a small helicopter. That expertise and resource was subsequently deployed to control turkeys and capture eagles. Use of capacity already on island for these other projects reduced the need to mobilise wholly separate efforts. It also made it possible to integrate the activities of the different projects and so reduce costs often encountered in projects like these that need to be implemented adaptively; teams often needed to wait for opportunities to engage that were unpredictable (e.g., some golden eagle capture strategies depended upon particular weather conditions and fortuitous sightings of birds).

Programmes designed to concurrently manage multiple pests can lead to greater effectiveness as well as efficiency. For example, as the pig eradication programme was transitioning from hunting to monitoring, some members of the pig hunting team were trained to identify priority pest plants, and tasked with mapping weeds island-wide using their helicopter, GPS, and database management expertise. The helicopter helped increase efficiency in mapping (Knapp *et al*. 2009) and in treating remote infestations that would have otherwise gone undetected or been difficult or unsafe to access. A co-benefit of this effort was that while conducting the weed work, the team also surveyed for pig sign – and so enhanced confidence that pigs had been eradicated. Ideally, synergistic activities like these that leverage limited funds to accomplish multiple restoration objectives would be built into programmes from the onset.

All that said, managers did have some constraints on their ability to sequence eradication efforts differently. For example, TNC was not authorised to control pigs until after it attained full property right in 1987. Moreover, the technological and methodological advances that today make concepts like concurrent multi-taxa eradication on an island of this size feasible were not yet established. Thus, this retrospective is not intended to critique decisions that were made. Rather, it is to take advantage of a vantage provided by multiple decades of eradication efforts to extract lessons that might inform future programmes.

DISCUSSION

The various pest management programmes on Santa Cruz Island have been essential to the protection of the island's unique native flora and fauna. That is not to say there have not been undesired or unanticipated effects. Given the degraded state of many islands and the complexity of their community dynamics, the unexpected should be expected. Planning must therefore be rigorous, and there needs to be strategic investment in monitoring so that risks to island resources can be identified and managed.

I acknowledge that this overview is largely anecdotal. There may be many hypotheses to explain apparent cascades, and causality can be difficult to determine (e.g., Bergstrom *et al*. 2009a, b; Dowding *et al*. 2009). The monitoring and experimentation necessary to demonstrate some of the trophic relationships discussed here were mostly absent. This lack of comprehensive monitoring is not atypical; it reflects the real resource constraints many island managers face. When action is imperative and funding is limited, an unfortunate trade-off is often research and monitoring. Fortunate for the conservation management of Santa Cruz Island is that the island is the focus of much ecological research, so the monitoring investments managers could afford were augmented by the work of external scientists who helped keep a pulse on the system and brought to light issues requiring management attention. This is important because cascades can play out over very long timeframes (e.g., Fig. 1) and anticipating the variables important to measure can be difficult. It would have been arguably impossible, for example, to predict that the presence of pigs would lead to the near extinction of foxes due to hyperpredation by a novel predator.

When there are gaps in monitoring, however, questions about effects of management actions can linger. For example, did the availability of carcasses during the sheep eradication on Santa Cruz Island and or the pig eradication on Santa Rosa Island provide the initial food subsidy that drew golden eagles to the island? Possibly, but the evidence suggests no. Many golden eagle nests on the islands have been excavated. Analyses of prey remains have not revealed sheep remains in nests from Santa Cruz Island or pig remains in those from Santa Rosa Island (Collins and Latta 2005). The more likely effect of sheep on the foxes was the destruction of vegetation cover that increased the exposure of the foxes to a novel aerial predator.

One strategy to reduce the risk of perverse outcomes is to leave fewer pests in the system that can go awry. Holistic pest management programmes may help reduce risk of perverse cascades (Zavaleta *et al*. 2001). But what also should be recognised are the efficiencies that can result from a comprehensive and strategically sequenced programme. If one pest plays a transformer role in the system, e.g., top down control of vegetation and therefore habitat quality for other pest species, that impact might be a means by which those other pest species can be managed more efficiently and effectively. Even those pests that seem relatively innocuous (like the small population of turkeys probably did before the fox crisis) might best be proactively engaged. Conclusive demonstration of adverse effects of pests should not be the threshold for intervention on an island: it was not known the extent to which, if at all, the turkeys would serve as a prey resource for golden eagles and exacerbate the risk to foxes. The precautionary principle was sufficient for action, as addressing pests early in their invasion may bring far fewer cascading consequences than doing so after a long period of ecosystem alteration.

Exploiting synergies among pest management projects might also improve the quality of the efforts relative to them being conducted separately. For example, the certification monitoring required at the end of the feral pig eradication contract was extensive (see Parkes *et al*. 2010). The hunters were obligated to search intensively for pigs despite the high likelihood none would be found. Months of such searching can strain morale. But by shifting the emphasis of the hunters – who by then were practically instinctually cued to see pig sign – to include other projects (like weed mapping), hunters were more focused in the field and managers were able to get both "products." The best demonstration of the benefit of synergistic activities was that the last pig dispatched on the NPS property was detected by the hunters while they were surveying for golden eagles – not pigs.

Fortunately, island managers today can benefit from considerable advances in eradication science and practice when planning to engage multiple pest problems. Eradication professionals have honed approaches to address many pest taxa such that coordinating efforts to engage multiple pests in an effectively single mobilisation may often be possible. With today's approaches, and adequate investment (e.g., in aerial support), it is conceivable that if we were presented again with a problem like Santa Cruz Island C. 1980, what took multiple decades might well have been completed in a few years – and for considerably less overall expense.

Exploiting trophic relationships among pests can be an important strategy for increasing the return on investment of limited conservation resources and increasing the pace and scale of island restoration (see Saunders *et al*. 2011). This case study suggests ways that pest eradication efforts might be strategically sequenced into more compressed and comprehensive programmes that will help manage complexity, reduce risk, and increase efficiency in meeting conservation goals. Enhancing the resilience of island ecosystems by effectively addressing multiple pest problems is imperative for the protection of many native species – especially in an era of increasing global change and uncertainty.

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Control and eradication of feral cats: field trials of a new toxin

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Abstract Feral cats (*Felis catus*) have caused the decline and extinction of threatened species on islands worldwide. The eradication or long-term control of cats is therefore an essential part of restoring native communities on these islands. In most situations, a combination of lethal techniques is required to remove feral cats, including trapping, hunting and poisoning. Para-aminopropiophenone (PAPP) is being developed as a new, humane poison for feral cats. Mammalian carnivore species appear more susceptible to PAPP than birds, so it potentially has higher target selectivity than other available toxins. A proprietary formulation of PAPP (PredaSTOP) developed by Connovation NZ Ltd. has been shown to kill cats humanely when delivered in a meat bait in pen trials. Two field trials of the formulation were undertaken with radio-collared cats. Toxic baiting was carried out by placing meat baits containing 80 mg PAPP in bait stations. Five of eight radio-collared cats in the South Island study and 13 of 16 radio-collared cats in the North Island study were poisoned. In the latter study, an additional three cats without collars that were monitored using infra-red cameras were also poisoned. Our results indicate that PAPP is an effective toxin for cats in the field, with potential application for their eradication or control on islands.

Keywords: *Felis catus*, field trial, New Zealand, humane, para-aminopropiophenone, PAPP, poison, radio-tracking

INTRODUCTION

Domestic cats (*Felis catus*) were brought to New Zealand from 1769 onwards and transported to many islands where they caused initial extinctions as well as ongoing declines of numerous threatened species (Dowding and Murphy 2001; Gillies and Fitzgerald 2005). Globally, the effects of cats on island vertebrates has been so severe, their eradication or control on some islands has become an essential part of preserving and restoring biodiversity (Courchamp *et al*. 2003; Nogales *et al*. 2004). In most situations, several lethal techniques are required to achieve cat eradication, including trapping, hunting and poisoning (Veitch 1985, 2001; Nogales *et al*. 2004). In a recent review of cat eradications on islands, toxic baits targeting cats were used in 31% of operations where the eradication methods were documented (Campbell *et al*. 2011).

Para-aminopropiophenone (PAPP) is being investigated as a new humane toxin for introduced predators, including feral cats, in both New Zealand and Australia (Marks *et al*. 2004; Fisher *et al*. 2005; Murphy *et al*. 2007; Johnston *et al*. 2011). Previous research on PAPP has explored its potential as a cyanide antidote (Baskin and Fricke 1992), as a radio-protective agent (DeFeo *et al*. 1972), and as a selective toxin for controlling coyotes (*Canis latrans*) (Savarie *et al*. 1983). The toxic effects of PAPP appear to be related to the rapid formation of methaemoglobin in some species. A high concentration of methaemoglobin leads to a rapid and lethal deficit of oxygen in cardiac muscle and the brain, resulting in animals becoming lethargic and unconscious prior to death (Vandenbelt *et al*. 1944; Marrs *et al*. 1991). PAPP has generally lower oral toxicity to birds than to mammalian carnivores, so presents some degree of target selectivity (Savarie *et al*. 1983; Fisher *et al*. 2008; Eason *et al*. 2010). PAPP is rapidly metabolised and excreted and is unlikely to cause secondary poisoning (Wood *et al*. 1991; Eason *et al*. 2010). Dogs (*Canis familiaris*), laboratory rats (*Rattus norvegicus*) and macaques monkeys (*Macaca fascicularis*) given sub-lethal doses of PAPP excreted 75-85% of it within 24 hours (Wood *et al*. 1991). Methylene blue (methylthioninium chloride) is a widely-available and effective antidote for methaemoglobinemia caused by PAPP poisoning (Bodansky and Gutman 1947).

A proprietary formulation of PAPP (PredaSTOP) has been developed by Connovation NZ Ltd. Feral cats fed 80 mg of PAPP in this formulation in meat baits became lethargic after 22-55 minutes, lost consciousness without spasms or convulsions and died after 54 to 125 minutes (Murphy *et al*. 2007). The aim of the study reported here was to determine the field efficacy of PredaSTOP in reducing feral cat numbers, to provide data for registration purposes.

MATERIALS AND METHODS

Study areas

The first trial was undertaken in May 2008 in the South Island, at two sites in North Canterbury: the Patoa pig farm near Culverden (c. 480 ha) and the Kate Valley landfill near Waipara (c. 100 ha). The second trial was undertaken in June 2009 at Ngamatea Station, between Taihape and Napier on the central plateau in the North Island. This site was C. 1500 ha of pasture, pine (*Pinus radiata*) windbreak hedging, and seral vegetation.

Radio tracking

Cats were trapped using Havahart live capture traps and were anaesthetised with intramuscularly injected Domitor (50-100 µg/kg) in the first trial, and Domitor (50-100 µg/kg) and Ketamine (100 mg/kg) in the second trial. Radio transmitters with an external whip aerial were attached to the cats using collars. After securing the collars in the first trial, the cats were injected intramuscularly with Antisedan (125-375 µg/kg) to reverse the anaesthesia until they were fully revived (*c*.10-30 min) and then released. In the second trial, cats were returned to covered cages to recover and released when fully revived (*c*.30-40 min). The radio transmitters (Sirtrack Ltd) emitted 40 pulses per minute with a 'mortality' function that switched to 80 pulses per minute after 12 hours without movement. Tracking was carried out using a TR4 (Telonics, Inc) receiver and a Yagi three-element aerial. Three infra-red motion-detection cameras (DigitalEye12.1 in IR Stealth Flash, Pixcontroller) were used in the second trial to monitor cats visiting bait stations. Cameras were moved around the study area and put at each station for at least two nights in the pre-feeding stage and were then used to check that cats returned to the stations after being radio-collared. Once this was confirmed, the cameras were used to monitor three cats without collars that were consistently identified visiting stations.

Poison baiting

PredaSTOP paste (200 mg) was applied to c.15 g meat baits to deliver 80 mg of PAPP per bait. Meat baits consisted of minced beef (trial 1) and minced rabbit (trial 2) in a ball around the PAPP paste. 'Submarine' bait stations (see Fig 2 in Warburton and Poutu 2002) were used in both trials to minimise non-target interference. Before toxic baiting, pre-feeding was carried out by removing the wire mesh from the ends of the bait stations and placing tracking cards inside. Once prints of cats were found in most feed stations the wire mesh was then attached to either end of the bait station, limiting access to the top entrance.

Trial 1: South Island

Twenty-two bait stations were spread around the pig farm and 10 bait stations were distributed at the landfill. There were three nights of PAPP baiting at the pig farm and eight nights at the landfill. Between one and three baits were placed in each bait station and checked each day to assess condition. Cats were radio-tracked daily to determine whether they were still alive and in the area.

Trial 2: North Island

Toxic baiting was carried out for five nights by placing five baits in each of 22 bait stations spread around the farm. Weather conditions were recorded and baits were checked, counted, removed each morning and replaced each evening. A snow storm on the fourth night meant that this night of baiting was delayed until the following night. As before, cats were radio-tracked daily.

RESULTS

Trial 1: South Island

Eleven cats were captured and radio collared; six were at the Patoa pig farm and five at the Kate Valley landfill. Of the six collared cats at the pig farm, one left the study area before toxic baiting, four were found dead after the first night of baiting, and the remaining cat survived. Four cats without collars were also found dead, three after the first night of baiting, and one after the second night.

Of the five cats collared at the Kate Valley landfill, two were found dead before toxic baiting and appeared to have been crushed by heavy machinery. Of the remaining three collared cats, one was found dead after the first night of baiting and the other two survived. One cat without a collar was found dead after the first night of baiting and a second cat without a collar was found dead after the second night. The additional nights of baiting at the landfill site did not increase mortality amongst the radio-collared cats.

All 11 cats found dead after PAPP baiting (5 radio-collared and 6 without collars) showed cyanosis around the mouth, consistent with poisoning by PAPP. Cats poisoned by PAPP in this trial ranged in weight from 1.31 to 3.35 kg.

Trial 2: North Island

Twenty-one cats were caught and radio-collared; one of these died and four left the study area before toxic baiting. Thirteen of the 16 cats that were alive and in the study area at the time of toxic baiting subsequently died (Table 1). The three cats without collars monitored by cameras were also found dead after toxic baiting. All 16 cats showed cyanosis around the mouth, consistent with poisoning by PAPP. Overall mortality was 0.84 (95% binomial confidence interval 0.60-0.97 for underlying mortality rate) assuming each cat had an equal probability of mortality. Cats poisoned by PAPP in this trial ranged in weight from 1.37 to 4.52 kg.

Over the five nights of toxic baiting there was confirmed bait take by feral cats on 23 occasions, with sixteen of these attributed to the radio-collared and camera-monitored cats. Unidentified cats were therefore also probably poisoned, as bait take and cat prints were recorded from seven bait

Table 1 Details on the feral cats monitored at Ngamatea Station during the poison trial, and their fates. Toxic baiting was carried out for five nights, using five baits in each of the 22 bait stations spread around the site.

Colour/distinctive marks	Sex	Transmitter	Weight (kg)	Fate/days after poison deployed
Black	Female	00	2.60	Died/Night 1
Tabby	Female	22	3.30	Died/Night 1
Tabby	Male	36	2.94	Died/Night 1
Tabby nicked ears	Female	14	2.95	Died/Night 1
Tabby white face	Female	30	3.52	Died/Night 1
Tabby	Female	28	3.32	Died/Night 1
Tabby	Female	66	2.48	Died/Night 1
Tabby white paws	Female	16	2.14	Died/Night 1
Tabby	Female	No collar	1.37	Died/Night 1
Tabby	Male	No collar	4.52	Died/Night 1
Black	Male	24	2.60	Died/ Night 2
Tabby white paws	Male	44	1.73	Died/ Night 2
Tabby	Male	34	2.74	Died/ Night 2
Tabby	Female	76	3.05	Died/ Night 2
Black white collar	Male	38	2.78	Died/ Night 3
Tabby	Male	No collar	1.41	Died/ Night 4
Tabby	Female	20	3.06	Alive
Tabby	Male	46	3.19	Alive
Black white collar	Female	12	3.00	Alive
Black	Female	8	3.01	Outside the trial area
Tabby	Male	32	4.03	Outside the trial area
Tabby	Male	10	4.43	Outside the trial area
Tabby	Male	84	5.75	Outside the trial area
Tabby	Female	88	1.35	Died before the trial began

stations where no carcasses were found. On four occasions, multiple baits in stations were not entirely eaten but a monitored cat was found dead in the vicinity each time.

DISCUSSION

Our results are the first from field trials of PAPP baits targeting feral cats in New Zealand. They support the results of earlier cage trials (Murphy *et al*. 2007), and suggest that PAPP is an effective new tool for feral cat control in the field. Cats also died from partly eaten baits, indicating that using multiple baits in stations could be an effective strategy to overcome the reluctance some cats may have about eating whole baits.

Although feral cats are naturally cautious and can be difficult to trap (Twyford *et al*. 2000; Veitch 1985, 2001), cameras showed that all 21 cats in the North Island trial fed regularly on non-toxic bait from the submarine stations before being captured and collared. Four of the collared cats left the trial area immediately after release, suggesting that these procedures may have changed their normal ranges and behaviour. Although the other cats remained in the area, their foraging behaviour may also have been affected by capture and an association with human presence, possibly explaining why three of them did not enter the bait stations after being collared. In operational poisoning using bait stations, without prior live-capture, a higher bait take and resulting mortality may be achieved.

Nogales *et al* (2004) recommended that feral cats should be routinely eradicated from islands where possible and that new techniques should be developed to do this. PAPP promises to be a useful addition to available tools for cat eradications, especially on larger islands and in the early stages of eradication. After trapping and hunting, the most frequently used technique for eradicating cats from islands is direct poisoning (Nogales *et a*l. 2004). Poisoning can be the most successful and effective technique for reducing the population quickly (Veitch 1985; Twyford *et al*. 2000). The most commonly used toxin for primary poisoning of cats is sodium monofluoroacetate (1080; Campbell *et al*. 2011). Although its use for island eradications of cats has been successful, the use of 1080 can be controversial; it has broad-spectrum toxicity to mammals and birds, and primary and secondary mortality of non-target species can therefore be a concern (Eason 2002; Weaver 2003).

Although mammalian carnivores were more susceptible to PAPP weight-for-weight than most bird species tested, there is some variability (Table 2). Also, as most birds weigh considerably less than cats, some bird species could still be at risk of poisoning if they ingest PAPP baits intended for feral cats (Murphy *et al*. 2005). In Australia, non-target testing has indicated some bandicoots (small marsupial mammals) and varanid lizards are highly susceptible to PAPP (S. Humphreys pers. comm.). Reptiles as a group may be vulnerable to the toxic effects of PAPP, as acetaminophen (paracetamol) is used for control of brown tree snakes (*Boiga irregularis*) on Guam (Savarie *et al*. 2001) and this compound, like PAPP, elevates methaemoglobin to lethal levels in some species. No evidence was found of any non-target species eating PAPP baits in our trials, and we believe the submarine bait stations we used help ensure targeted delivery in our situation.

Other methods are being tested for delivering PAPP to feral cats (and other pests). One example is a tunnel system that uses compressed gas to propel a measured amount of PAPP paste onto the abdomen of pests as they pass over a trigger. Animals become exposed to the paste when they groom their coat. Cage trials have achieved proof of concept for this method as a means of killing stoats (*Mustela erminea*), indicating that a device capable

Table 2 Reported oral LD_{50} values (the dose required to kill 50% of the sample population) for PAPP.

Species	LD_{50} (mg/kg)	Reference
Domestic cat (*Felis catus*)	5.6	Savarie *et al*. 1983
Coyote (*Canis latrans*)	5.6	Savarie *et al*. 1983
Dog (*Canis familiaris*)	7.5	Coleman *et al*. 1960
Stoat (*Mustela erminea*)	9.3	Fisher *et al*. 2005
Bobcat (*Lynx rufus*)	10	Savarie *et al*. 1983
Kit fox (Vulpes velox)	14.1	Savarie *et al*. 1983
Ferret (*Mustela furo*)	15.5	Fisher & O'Connor 2007
Red fox (*Vulpes vulpes*)	< 25.2	Marks *et al*. 2004
Dama wallaby (*Macropus eugenii*)	89	Fisher *et al*. 2008
Badger (*Taxidea taxus*)	c. 100	Savarie *et al*. 1983
Raccoon (*Procyon lotor*)	142	Savarie *et al*. 1983
Rat (*Rattus norvegicus*, albino)	177	Savarie *et al*. 1983
Mouse (*Mus musculus*, albino)	223	Savarie *et al*. 1983
Striped skunk (*Mephitus mephitus*)	> 400	Savarie *et al*. 1983
Brushtail possum (*Trichosurus vulpecula*)	≥ 500	Fisher *et al*. 2008
Guinea pig (*Cavellio porcinus*)	1020	Scawin *et al*. 1984
Mallard duck (*Anas platyrhynchos* Pekin breed)	32	Eason *et al*. 2010
Mallard duck (*Anas platyrhynchos* Pekin breed)	38	Fisher *et al*. 2008
Red-winged blackbird (*Agelaius phoenicus*)	133	Savarie *et al*. 1983
Blackbird (*Turdus merula*)	174	Eason *et al*. 2010
Black-billed magpie (*Pica pica*)	178	Savarie *et al*. 1983
Common crow (*Corvus brachyrhynchos*)	≥ 178	Savarie *et al*. 1983
Coturnix quail (*Coturnix coturnix*)	> 316	Savarie *et al*. 1983
Starling (*Sturnus vulgaris*)	> 316	Savarie *et al*. 1983
Weka (*Gallirallus australis*)	568	Eason *et al*. 2010
Australian magpie (*Gymnorhina tibicen*)	1388	Eason *et al*. 2010

of safely delivering multiple lethal doses of toxin without regular resetting can be produced (Connovation NZ Ltd., unpubl. data).

In conclusion, potential non-target issues for PAPP should be lessened by the development of targeted delivery systems, such as bait stations, tunnel systems, or by specific bait presentations that exploit cat feeding behaviour and physiology (Marks *et al*. 2004; Marks *et al*. 2006; Johnston *et al*. 2011). Few toxins are currently available for the control or eradication of cats. We believe the development of PAPP represents a significant advance. It is humane in comparison to available toxins, more toxic to cats than birds, and presents a low risk of secondary poisoning.

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Removal of the American bullfrog *Rana (Lithobates) catesbeiana* from a pond and a lake on Vancouver Island, British Columbia, Canada

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Abstract The American bullfrog is listed as one of the 100 Worst Alien Invasive Species internationally because it is adaptable, prolific, competitively exclusive, loud, and predatory. An expectation of profits from the sale of frog legs for human consumption has led to bullfrogs becoming established on most continents as well as on islands in western Canada and the western United States, Hawaii, throughout the Caribbean, Crete, Indonesia, Japan, Singapore, Sri Lanka, and Taiwan. The ecological impact of bullfrogs on islands can be profound especially where ecologically vital freshwater resources may be limited. While the problems created by bullfrogs are well-documented, there have been few technological advances in their effective control and management. In 2006, a programme was initiated to design, field test, and refine new equipment and tactics to capture individual bullfrogs at rates to exceed replacement. The programme also hoped to demonstrate that bullfrog eradication is a feasible and practical option. The principal manual capture technique is modified fisheries electro-shocking tailored specifically for capturing juvenile (<80 mm body length) and adult (>80 mm body length) bullfrogs. Bullfrog tadpoles are not hunted directly but collected as they reach the latter stages of metamorphosis or have recently transformed. Clear patterns have emerged from comparative data sets collected between 2007 and 2009 that identify some basic units of bullfrog eradication, including logistical and time sequence requirements for successful removal of all age-classes from a single lake or pond after only one successful spawning. The two case studies presented here illustrate patterns useful for interpreting catch results and for predicting the time, effort, and costs in carrying out complete site eradications. In both examples, 'site eradication', i.e. reducing numbers of all bullfrog age-classes at one site from hundreds or thousands to zero, was carried out by one two-person team and achieved over three years with only a few nights effort per site per year. The cost of running this programme is currently \$400/night/2-person team. At Amy's Pond (0.4 km perimeter distance), 1587 adult and juvenile bullfrogs were collected after 23 nights of effort spread over 3 years for a total cost of CAN\$9200. At Glen Lake (2 km perimeter distance), 1774 bullfrogs were collected after 41 nights of effort spread over 3 years for a total cost of CAN\$16,000.

Keywords: Amphibian management, eradication, control, site eradication, electro-frogging, cost-effective

INTRODUCTION

Populations of alien invasive American bullfrogs, (*Rana (Lithobates) catesbeiana),* are now established in western North America, western Europe, south and east Asia, and Central and South America. Historically, live bullfrogs were exported from their native range in eastern North America to establish new wild populations supplying international markets for frog meat. Bullfrogs acclimatise readily to habitats ranging from temperate to tropical. Rapid population growth rates coupled with migration outward from source population leads eventually to bullfrogs in all habitable lakes and ponds. The result is potentially catastrophic for native species that are prey to this large, abundant and aggressive non-native predator. Eradication of bullfrog populations has been proposed out of concern for the sustainability of native ecosystems and species diversity, but also because of human objections to the noise produced by choruses of large male bullfrogs and their consequent effects on property values. Continental bullfrog populations can spread out geographically over wide areas. However, island populations are area-constrained, often with relatively few vital freshwater spawning 'sites' available and surrounding habitat that is bounded on all sides by a barrier of saltwater. Islands therefore have advantages if bullfrog eradication is to be attempted. Once eradication is achieved, islands should also be easier to keep bullfrog-free.

Vancouver Island is the largest island on the west coast of North America (32,134 km^2). Its cool mountainous interior, vast tracts of rocky terrain and thick forest restrict or inhibit bullfrog dispersal. However, bullfrogs have been released and are spreading from multiple disjunct pocket populations along the low, warm, coastal zone of south-eastern Vancouver Island. They have also been introduced to smaller, adjacent islands, and have for many decades populated regional Vancouver on the adjacent mainland coast (Fig. 1).

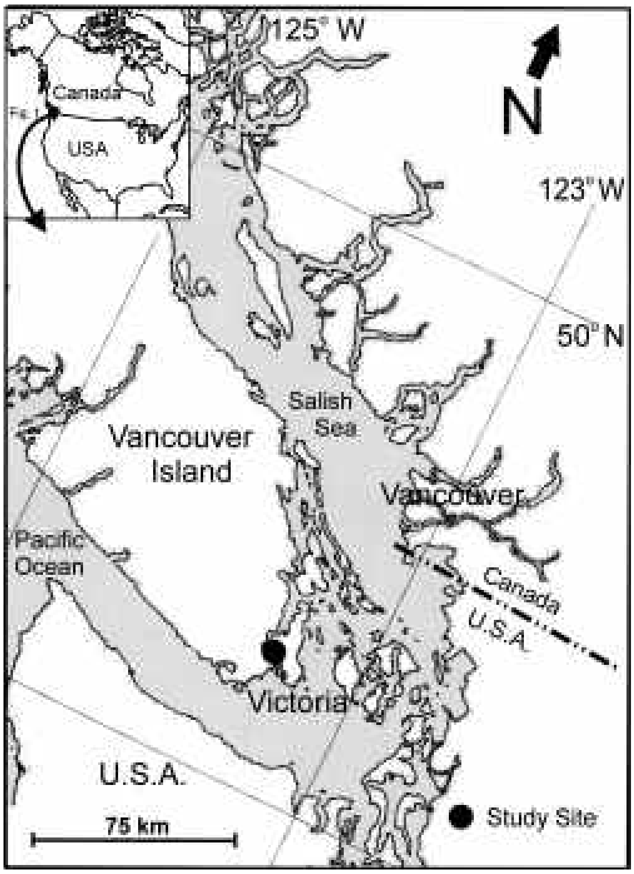

Fig. 1 Location of case study sites on the Saanich Peninsula, Vancouver Island, British Columbia, Canada.

Pages 217-221 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

There are few published case studies of bullfrog eradication, and the few successful examples were laborious and costly (Adams and Pearl 2007; Kraus 2009). In England in 1996, the eradication of bullfrogs from only a few small ponds cost approximately US$70,000, including the earth-moving equipment that ultimately destroyed freshwater habitat (Banks *et al* 2000; CABI Bioscience 2005). In Germany between 2001 and 2004, bullfrogs were eradicated from five ponds with help from a volunteer force of 20 as well as the local fire department and an 'electro-fish' team. Cost estimates for this project were US$80,230/pond/year for five ponds or US$409,000 annually (Reinhardt *et al* 2003; Nehring and Klingenstein 2008). These European case studies utilised large work forces and heavy equipment beyond the budgets of many agencies. Other attempts at managing or eradicating invasive bullfrog populations have used netting, barrier fencing, seining, shooting, gigging (spearing), pitfall traps, and pond draining. These technologically unsophisticated attempts have been mostly ineffectual, excessively labour-intensive, and unable to keep pace with the bullfrogs' prolific reproduction and mobility. Such attempts are particularly difficult where populations have grown to maturity and have dispersed geographically before any control efforts were attempted. A general impression is then formed that bullfrog eradication may be feasible through the intense countervailing efforts of a large and dedicated workforce, but the time-consuming exertions required also make these measures exorbitantly expensive and generally impractical (Adams and Pearl 2007; Krause 2009).

In this paper I describe cost-effectiveness of methods used to remove bullfrogs from a pond and a lake on Vancouver Island, British Columbia, Canada. For the purposes of this study, I use the following definitions:

A 'bullfrog site' is a discrete body of standing water – generally a lake, pond, or pool – where some or all life stages of bullfrogs are present. When all sites are identified regionally and brought 'under control' by the eradication programme then eradication is inevitable because standing water is vital for population sustainability and growth.

'Productive sites' have the essential elements of: 1) permanent water that does not freeze to the bottom of become anoxic in winter; and 2) summer surface temperatures that reach and exceed 25° C. for an interval of weeks in mid- to late summer to facilitate reproduction. Permanent water is a requirement because, at this latitude, bullfrog tadpoles will commonly take 24 to 36 months to reach metamorphosis.

'Non-productive sites' are either: 1) impermanent pools that trap and kill bullfrog tadpoles before they metamorphose; or 2) too cool in summer for reproduction to occur, e.g., <25° C. Non-productive sites are useful only to migrating bullfrogs as way stations or as over-wintering sites.

STUDY SITES

The two case studies presented here are drawn from preliminary results of a long-term regional control program that encompasses a cluster of lakes and ponds at the isthmus of the Saanich Peninsula, at the extreme southern end of Vancouver Island, including the City of Victoria (Fig. 2). The particular significance of the case studies presented is that the sites are dissimilar in size and habitat characteristics, but comparable in their stage of bullfrog colonization. In both instances, fieldwork began shortly after the arrival of adult bullfrogs and after one spawning had occurred at each site. It was unknown at the start how many tadpoles would reach metamorphosis and how much time and effort would be required to capture them all post-transformation. The innovative manual capture technique developed specifically for this program was, at that stage, untested. At the end of the third field season (2007 – 2009) it was possible to quantify material costs, time and effort required to de-populate both sites using the 'electro-frogger' technique.

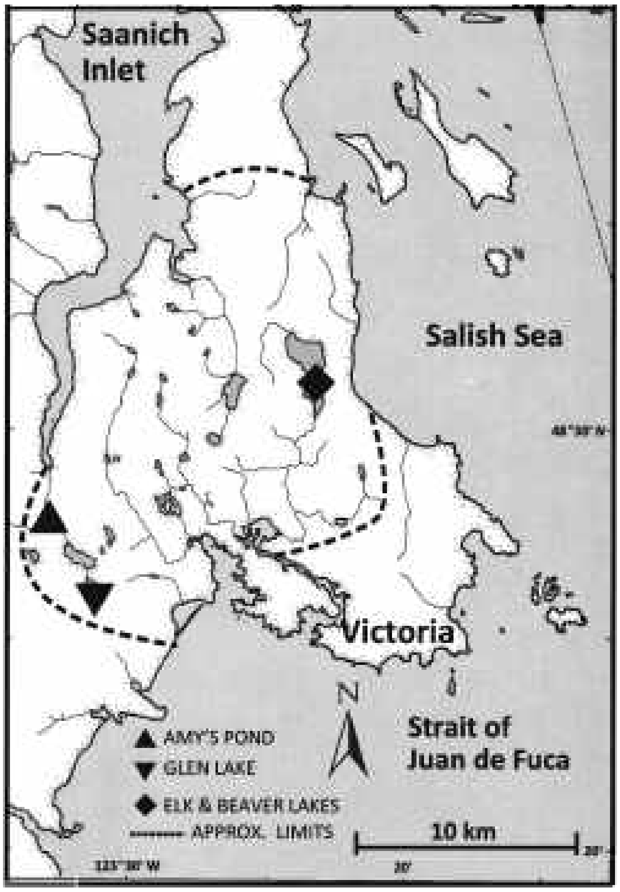

Fig. 2 Site of the founding bullfrog population (diamond) and current approximate distribution limits of bullfrogs on the Saanich Peninsula, British Columbia, including the case study sites Amy's Pond and Glen Lake.

1. Amy's Pond

At Amy's Pond the margins were essentially bare of aquatic and emergent vegetation throughout the summer. This meant that despite somewhat turbid water, there was good visibility at the surface and accessibility to the margins. With a perimeter distance of only 0.4 km, many circuits of Amy's Pond could be made in a single three-hour evening session and virtually every individual of every post-larval age-class present could be located and captured on any given night.

2. Glen Lake

Glen Lake had a perimeter distance of about 2 km, or five times the margin of Amy's Pond. It was also much more florally complex with many species of aquatic, floating, and emergent plants, as well as riparian shrub and tree thickets. These all provided effective cover for bullfrogs, impeded vision during searches, and interfered with the ability to manoeuvre during approach and capture. Unlike at Amy's Pond, only one thorough circuit of Glen Lake could be completed per evening and this only when bullfrog numbers were very low. While bullfrog densities were high, only a portion of the lake margin could be cleared per evening session.

MATERIALS AND METHODS

For this programme, one two-person team is the minimum manpower unit so what follows are the requirements to equip, transport, and fund one team. Transportation includes a utility vehicle and a very sturdy inflatable rowboat. Essential field equipment includes a modified fisheries electro-shocker, 'electro-frogger' pole, powerful spotlights, and two chest freezers, with one modified to maintain a temperature slightly above freezing. The freezers were used in a two step euthanasia procedure.

On southern Vancouver Island, the field season began in April and ended around the beginning of October. Fieldwork was weather-dependent and incompatible with excessive wind (> 15 km/hr) or rain. As explained, the case studies are part of a larger regional programme that encompassed many more sites. Regionally, we worked every night with suitable weather, which amounted to 93 nights in 2007 (19 sites/4,479 bullfrogs), 114 nights in 2008 (20 sites/3,430 bullfrogs), and 125 nights in 2009 (28 sites/3872 bullfrogs). Costs averaged about $400/night/team or CAN$37,200 in 2007, CAN$45,600 in 2008, and CAN$50,000 in 2009. The programme also included daytime site assessments, examination and measurement of the catch, dissections, data compilation and analysis, and write-up of results. On-going annual maintenance costs included permits and licences, liability insurance, and automobile insurance, as well as routine costs such as fuel, facilities, utilities, website, public relations and equipment repair and replacement.

In 2006, a prototype electrode-fitted pole (electro-frogger) was developed and field tested, and more refined, patent-pending versions have been employed since 2007. During the summers of 2007 to 2009, a two-person team applied this manual capture technique for four-hour sessions on every evening that weather permitted. A four-hour session included loading and unloading equipment, so the time locating and capturing bullfrogs was approximately three hours. Teams worked at night from an inflatable boat, with one person to manoeuvre and position the boat while the second person located and caught juveniles (< 80 mm body length) and adults (> 80 mm) frogs. Pond and lake margins were scanned by spotlight to detect bullfrogs by their eye reflections. Vocalisations from adult male bullfrogs also independently identified their whereabouts. Bullfrogs were dazzled and transfixed by the spotlight's beam as we approached. Then the electrode-fitted pole was used to generate a subsurface concentrated electrical field of < 50 cm diameter near the target bullfrog. The electrical field stunned and temporarily paralysed juvenile and adult bullfrogs for 30 seconds to one minute, which was enough time to get them into a container. The technique is humane, species-specific and only targets one bullfrog or small groups of bullfrogs in very close proximity to one another. Capture rates, on any given night, are influenced by each site's habitat characteristics, weather, and bullfrog density and demographics.

For euthanasia, bullfrogs were placed into a chest freezer modified to lower their core body temperature to just below 2° C. After at least 12 hours they are transferred to a conventional deepfreeze that quick-freezes the now cold-stupified bullfrogs. They remain in the second freezer for at least 48 hours. Cold is a natural anaesthetic for amphibians and freezing leaves an uncontaminated, chemical-free carcass that can be safely used to feed injured wildlife, donated to high schools for educational dissections, or composted.

RESULTS

In the spring of 2007, Amy's Pond and Glen Lake were at the same initial stages of bullfrog colonisation. At Amy's Pond, few adults were present, there were a few new arrivals, and there had been one successful spawning 12 to 24 months previously, which produced many tadpoles. Around mid-summer 2007, this single cohort of bullfrog tadpoles began to metamorphose and on 30 August we collected 237 transforming or recently transformed juveniles and five adults. Transformations continued throughout the remainder of the summer, but the number of juveniles captured per evening declined markedly with each subsequent visit in 2007 (Fig. 3a).

Fieldwork re-commenced in April 2008 (Fig. 3b) as the over-wintered remnant of the same cohort became active and began to complete their transformations. By the end of the 2008 season, we could find no bullfrogs of any age-class.

Our 2009 results confirmed that the metamorphosis event that began mid-summer 2007 was essentially over by mid-summer 2008. Spawning was prevented from 2007 onward by clearing the pond of all adults prior to the mid- to late-summer spawning period. By 2009, Amy's Pond was tadpole-free, though there was a small but persistent influx of juveniles and young adults from adjacent lakes and ponds.

Ultimately, we removed 1587 bullfrogs from Amy's Pond by investing 3 hours of collecting effort in each of 23 nights spread over 3 consecutive summers. By the end of the 2008 season, bullfrog numbers had been reduced to zero and all bullfrogs encountered thereafter were the result of immigration or release. The total cost for this three-year (23 nights) effort was CAN$9200 (Table 1).

Like Amy's Pond, Glen Lake was in the earliest stage of bullfrog colonisation in 2007 with just one successful spawning. By mid-summer 2007, bullfrog tadpoles first noted in late-2006 had begun to metamorphose. On 25 July, we collected 59 bullfrogs (Fig 4a), all but one of which was either in the latter stages of metamorphosis or had just recently completed transformation. From 25 July to 16 August, we concentrated on one end of the lake where the number of juveniles was high and the conditions were especially difficult due to extensive patches of cattail, rushes, water lilies, various floating aquatic plants, and willow thickets. By 17 August, one end of the lake was clear of bullfrogs and efforts were moved to the opposing end, which was also heavily vegetated. Tadpole metamorphosis followed a pattern similar to Amy's Pond, commencing in mid-summer 2007 with transformations continuing throughout that summer (Figs. 3a, 4a).

Table 1 Comparison of site characteristics with time and cost of achieving 'site eradication'

Sites	Perimeter	Littoral/ Riparian	Nights/year	Catch/year	Cost/year	3-year total catch/cost
Amy's Pond	0.4 km	Florally barren	8/2007 10/2008 5/2009	871 661 55	$3200 $4000 $2000	1587/$9200
Glen Lake	2.0 km	Florally abundant & complex	16/2007 16/2008 9/2009	1376 366 32	$6400 $6400 $3600	1774/$16,400

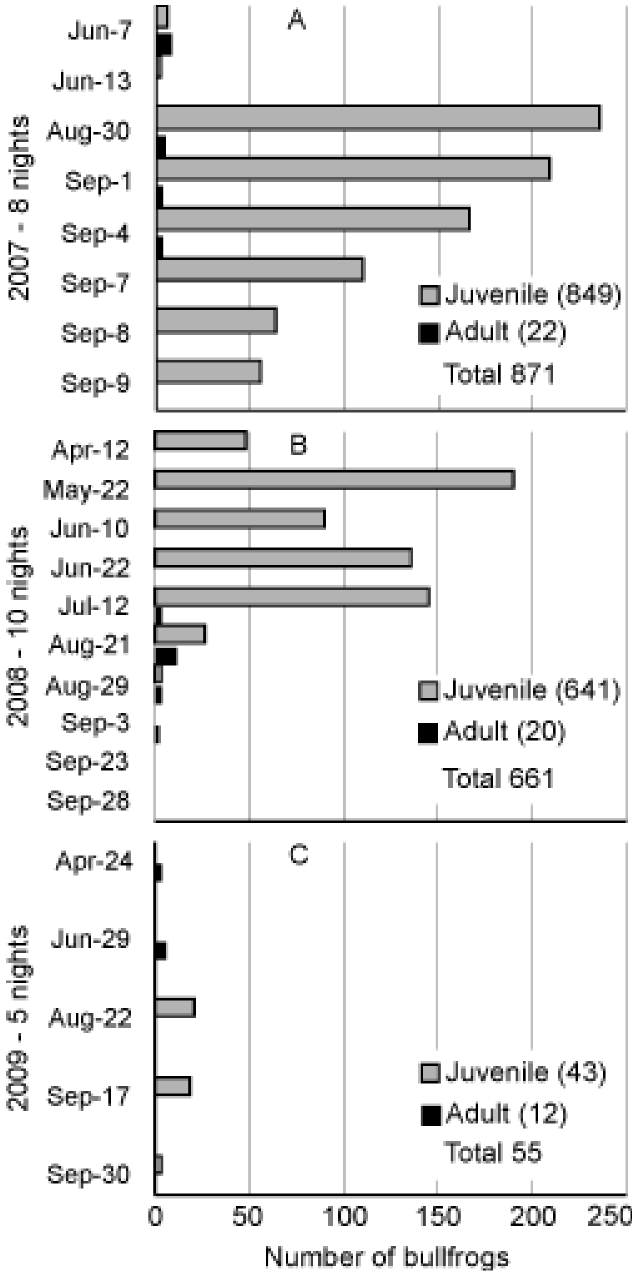

Fig. 3 Amy's Pond chronology and nightly capture results 2007- 2009 (n = 1587).

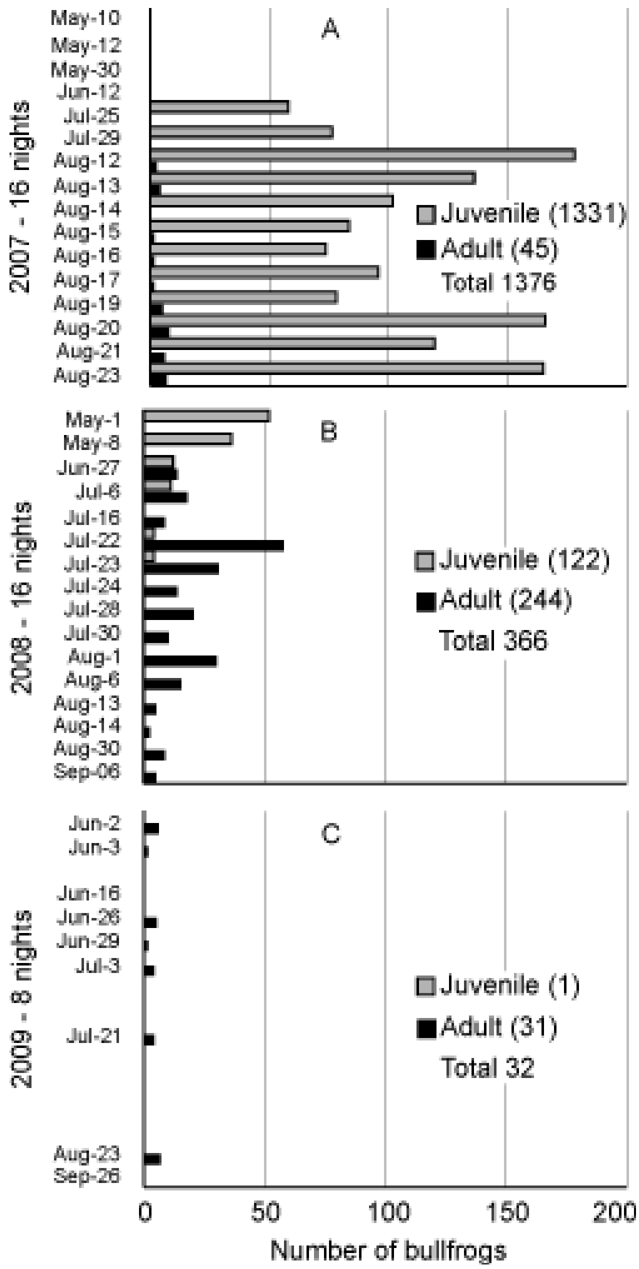

Fig. 4 Glen Lake chronology and nightly capture results 2007 - 2009 (n = 1774).

The 2008 season (Fig. 4b) began with a resumption of metamorphosis that tapered off to near zero by mid-summer. Adults recorded from 27 June onward undoubtedly included a few immigrants but were primarily Glen Lake juveniles whose body lengths had grown rapidly to young adult size (>80 mm body length) before we were able to locate and capture them.

In 2009, there were only a few newly arriving adults and juveniles. Total costs for this three-year (41 nights) effort was CAN$16,400 (Table 1).

DISCUSSION

By the end of the 2009 field season, all age-classes of bullfrogs had been successfully removed from both sites. Excluding repopulation through natural immigration or human translocation, both Amy's Pond and Glen Lake were then free of bullfrogs.

The two case studies are comparable because both had only one spawning per site. Without knowing how many eggs were produced by each of the two adult females there was nevertheless remarkable similarity in the timing and interval of tadpole transformation, and in the numbers of metamorphs/juveniles ultimately captured. If it is assumed that each female produced thousands of eggs, then there must have been considerable mortality in the tadpole stage to have resulted in only about 1,500 metamorphs/juveniles taken from each site. This is one reason to ignore the tadpole stage and concentrate on capturing the post-metamorphic stages if tadpole mortality is consistently high.

Another similarity between these case study results is a pattern of asynchronous cohort transformations from tadpole to juvenile that stretches over 12 months and two calendar years. For example, for each cohort there was an induction stage to this incremental metamorphosis that commenced about mid-summer of one year and continued throughout the remainder of the active season, e.g., July to October. However, some of this tadpole cohort did not metamorphose before the onset of winter, completing transformation the following spring in a protracted conclusion stage, e.g., April to August that peaked in spring. If this pattern proves to be consistent, a manual capture technique that targets only post-metamorphic stages will, by necessity, require two calendar years or more to clear a

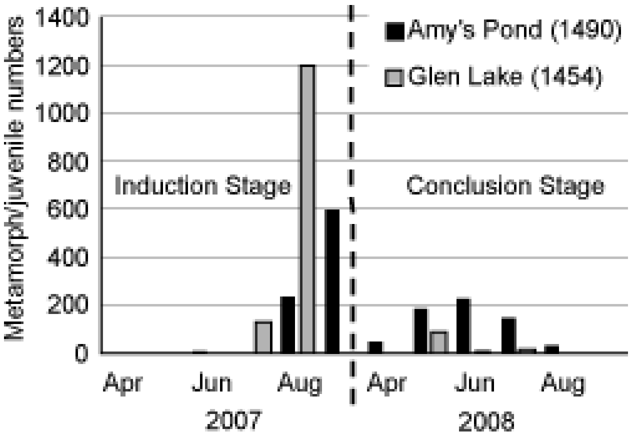

Fig. 5 Comparative capture results of the metamorph/ juvenile size-classes (<80 mm body length) from Amy's Pond and Glen Lake. Both sites exhibited a 2-stage incremental cohort metamorphosis.

lake or pond of all bullfrogs. If spawning has occurred in two or more consecutive years then the removal process will take three or more calendar years to complete. At Amy's Pond, 57% (849) of our 2-year total of 1490 metamorphs/ juveniles were captured during the induction stage in 2007 and the remaining 43% (641) during the conclusion stage in 2008. In Glen Lake, 92% (1332) of our 2-year total of 1454 metamorphs/juveniles were captured during the induction stage in 2007 and the remaining 8% (122) during the conclusion stage in 2008 (Fig. 5).

The electro-frogger manual capture technique demonstrated a capacity to collect as many as 241 bullfrogs per three-hour session at Amy's Pond and 181 per three-hour session at Glen Lake (Fig. 3, 4).

CONCLUSIONS

1. The manual capture 'electro-frogger' technique, when competently and diligently applied and when coupled with various pieces of essential accessory equipment, successfully located and captured juvenile and adult bullfrogs at rates that far exceeded replacement.

2. The 'electro-frogger' does not place all individuals of the population at risk simultaneously because the tadpole stage is largely unaffected. However, as tadpoles transform from landlocked aquatic larvae to semi-aquatic juveniles they rise to the surface and become vulnerable to capture.

3. At the latitude of Vancouver Island, adult bullfrogs can be successfully located and removed as they emerge from winter torpor (April – May) and prior to the spawning season (July – September). This means that with appropriate intensity of effort, bullfrog reproduction can be prevented within the first few weeks of the first year of an eradication programme and similarly prevented in subsequent years.

4. A singe two-person team can eradication bullfrogs from small to medium-sized water bodies but the number of nights per year required per year will vary depending upon perimeter distance and habitat characteristics at each site as well as the age-class complexity of the bullfrog population. An additional team would not have reduced the number of nights or number of years required to bring Amy's Pond under control. However, the number of nights per year spent on the much larger Glen Lake would have been significantly reduced by adding a second team. The number of years, however, remains independent of the number of teams deployed since each cohort of tadpoles begins to metamorphose in one calendar year and finishes in the next.

5. Where bullfrogs have spawned more than once in the same year, at the same site, the number of resultant juveniles will be numerically greater than reported here. However, they can still be removed within two years from the onset of metamorphosis if sufficient effort is applied in terms of increasing the number of field nights per year and/ or increasing the number of teams active per site per night. Where there has been multiple spawning in each of two or more consecutive years, then it will take three to four years to achieve the same result with appropriate proportional increases in the intensity of effort.

6. The case studies presented here represent an environmental situation characteristic of a particular latitudinal range and climatic regime. Results from southern British Columbia should be directly relevant to bullfrog invasions in Europe, northern Asia, western United States, and possibly southern South America. It would be helpful to have comparative data sets from subtropical and tropical regions where bullfrogs are active year-round and the tadpoles reach metamorphosis within 12 months. Conceivably, a comparable programme in warmer climates with no winter dormant period would move along much faster than in these case studies, in which case site eradication through manual electro-frogging may be achievable in as little as 12 months.

7. The proposition that bullfrog eradication is neither feasible nor practical is contradicted by this study. Furthermore, the technique used is time-efficient, cost-effective, humane, and safe for personnel and the environment.

ACKNOWLEDGEMENTS

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A summary of the current progress toward eradication of the Mexican gray squirrel (*Sciurus aureogaster* F. Cuvier, 1829) from Biscayne National Park, Florida, USA

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ABSTRACT In 2007, the US National Park Service (NPS) began an eradication programme against Mexican gray squirrels (*Sciurus aureogaster*) on islands within Biscayne National Park. Planning included the 2007 development of a management plan, efficacy monitoring, a resource management weapons training programme, and ground and aerial surveys to locate dreys (squirrel nests) for follow-on removal. Aerial survey for dreys was incorporated in 2008 and included the use of helicopters and a digital aerial sketch mapping technique. During eradication operations, marked dreys were systematically visited after dusk by trained technicians on foot. Each drey was precisely destroyed by a shotgun using non-lead ammunition and humanely euthanasing any occupants. Project monitoring consisted of regularly scheduled aerial and ground drey surveys, camera traps and nest boxes. Since eradication operations began in 2007, 1410 dreys have been located, marked, and removed from 1360 trees. A total of 33 squirrels have been removed (15 male and 18 female) from Elliott and Sands keys. The eradication project has been a collateral duty of NPS biologists and has been conducted as funding and staff time permit. In light of this limitation, the project is ongoing with complete eradication expected in early 2011. Current project cost is approximately $US70,000 and the final cost is estimated to be $US80,000.

Keywords: Squirrel invasions, rodent eradications, restoration, dreys, nest boxes

INTRODUCTION

Rodents have been eradicated from over 332 islands around the world (Howald *et al*. 2007), often with significant benefits to native biodiversity (e.g., Rauzon 2007). Of the mammalian invaders, rodents present formidable ecological and economic threats, which are exemplified by tree squirrels (Palmer *et al*. 2007). Biological characteristics that have enabled tree squirrels to become invasive include: high reproductive potential, high vagility, diverse food habits, ability to construct nests, and plasticity in human-impacted landscapes. Islands are particularly vulnerable to these invasions because tree squirrels are also able to establish viable populations with very small propagules (Palmer *et al*. 2007). At least two large-scale squirrel eradication attempts in Europe have failed. In Great Britain during the 1940s and 1950s, attempts to eradicate *Scuirus carolinensis* included private citizen hunting efforts (Sheail 1999) and poisoning (Dagnall *et al*. 1998; Sheail 1999), which led to dissent from animal rights groups. This case did give rise to new ideas about squirrel control including manipulation of the physical environment and sterilisation (Dagnall *et al*. 1998). The second attempt against *S. carolinensis* was in Italy but was halted because of protests from animal rights groups (Bertolino and Genovesi 2003).

In this paper we outline an eradication campaign against the Mexican gray or red-bellied squirrel (*Scuirus aureogaster*) from islands in Biscayne National Park in Florida, USA. We describe the invasion, effects of squirrels on native species, methods used to delimit the populations, and their removal.

SQUIRREL INVASION

The Mexican gray squirrel is an arboreal species native to southern Mexico. Two pairs of squirrels were purposefully introduced from eastern Mexico to Elliott Key in Biscayne National Park in 1938 (Fig. 1), where they established and became widespread by the 1960s. Squirrels were also reported on the adjacent Adams Key and Sands Key and one was captured swimming across Caesar's Creek toward Old Rhodes Key (Layne 1997). The squirrels were considered extirpated in 1992 (Layne 1997), when the tidal surge from Hurricane Andrew submersed the islands (Ogden 1992; Davis *et al*. 1993). However, the species was subsequently found on Elliott Key indicating that a population had survived (Koprowski *et al*. 2005).

A survey of Elliot Key in 2005-2007 by Geoffrey Palmer of the University of Arizona focussed on the conspicuous leaf and stick nests (dreys) built by squirrels near the tops of trees as a refuge from weather, predators,

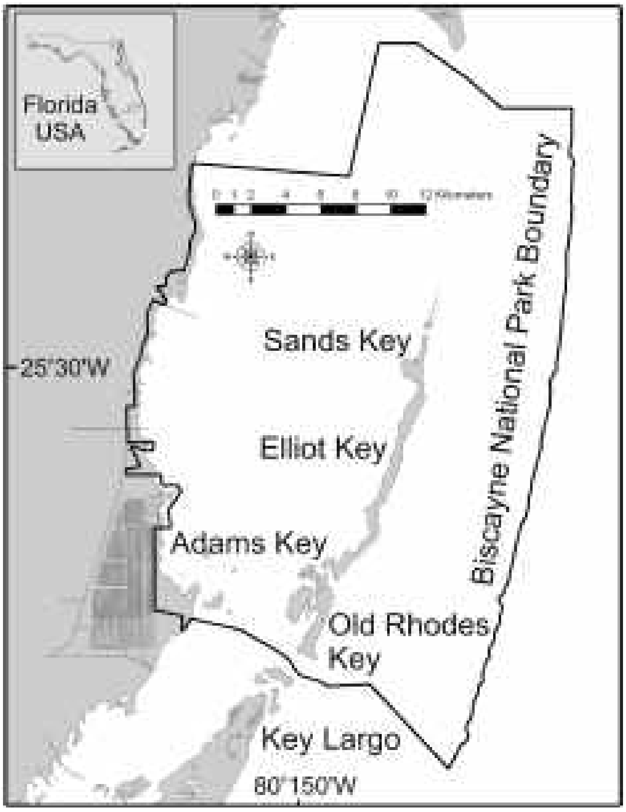

Fig. 1 Biscayne National Park and the keys named in the text.

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and as a safe place to rest and sleep (Brown and McGuire 1975). The survey revealed squirrels throughout the hardwood hammock forest habitat, with 115 nests (dreys) documented initially and more than 200 dreys documented over the course of the study (Palmer 2010). This survey was also conducted on other islands within the previous range of the squirrels, including those they had attempted to reach. Monthly surveys on Adams Key from December 2005 to July 2006 failed to find any sign of squirrels but squirrels were found on Sands Key and Old Rhodes Key in March 2007. This discovery raised further concern about the likely effects of this invasive species within and outside Biscayne National Park.

In Biscayne National Park, male squirrels had a home range of 2.3 ha, and females a home range of 0.9 ha (Brown and McGuire 1975). The squirrels breed year-round and are opportunistic feeders (Koprowski pers. comm.), relying heavily on introduced plants for food including coconut palm (*Cocos nucifera*), sapodilla (*Manikara zapota*), Australian pine (*Casuarina equisetifolia*), and papaya (*Papaya carica*) (Brown and McGuire 1975). Subsequent control of these and other non-native plants on Elliott Key has forced squirrels to rely on native plants for food, including the fruits of sea grape (*Coccoloba uvifera*), mastic (*Mastichodendron foetidissimum*), gumbo limbo (*Bursera simaruba*), keys thatch palm (*Thrinax morrissii*), Florida thatch palm (*Thrinax radiata*), and the endangered Sergeant's Buccaneer palm (*Pseudophoenix sargentii*). The squirrels also feed on birds' eggs and invertebrates. National Park Service assessments of effects of the squirrel on Elliott Key before Hurricane Andrew (Tilmant 1980) suggested that they preyed on the declining liguus tree snail (*Liguus fasciatus)* and collected palm leaves from the *Thrina*x spp. to line their nests. In 2006, Palmer (2010) found squirrels using parts of *Thrinax* to line nest cups, but failed to document any nests that utilised parts of the state endangered *Pseudophoenix sargentii.*

Other damage to native trees from squirrels included clipped branches and feeding on the plants' phloem, fruits and seeds. Nest trees were damaged during the construction and maintenance of nests as these trees were the primary source of nesting material. These data on range, foraging, and nest building by squirrels helped with the development of alternatives for conducting the eradication and was incorporated into a formal management plan for the species. The documented impacts on native vegetation from the squirrels strengthened the case for their eradication from Biscayne National Park.

The impact of the Mexican gray squirrel on South Florida ecosystems is poorly understood, although introduced populations of other squirrels throughout the world are known to have detrimental effects (Koprowski pers. comm.).

Primary concerns about the spread of squirrels within Biscayne National Park included: damage to native vegetation, such as the endangered *P. sargentii*, and state-threatened thatch palms *(Thrinax radiata* and *T. morrisii)*; avian nest predation; competition with the state-threatened white-crowned pigeon (*Columba leucocephala*); and feeding on the liguus tree snail, a species of special concern in Florida.

The potential for further spread of the squirrels to other islands and mainland Florida is of environmental, agricultural, and economic concern. Squirrels could compete with the federally endangered *Neotoma floridana smalli*, the federally endangered Key Largo cotton mouse (*Peromyscus gossypinus allapaticola*), the state threatened Big Cypress fox squirrel (*Sciurus niger avicennia*), the grey squirrel *(Sciurus caroliniensis*) and other native species. Potential damage to Florida's agriculture and tropical fruit production was also of concern, since Mexican gray squirrels are known to damage agricultural crops such as corn in their native range (Romero-Balderas *et al*. 2006).

The invasive potential of these squirrels was demonstrated from the aerial and ground surveys of dreys on Elliott Key. However, the isolation of these populations from mainland Florida suggested that the species could be eradicated. Primary goals for the eradication from Biscayne National Park included: 1) eliminate potential effects of the squirrels on natural resources within the Park; 2) remove any possibility for squirrel populations to expand their range outside of the Park; and, 3) increase public and agency awareness of the significant threats from invasive species.

MATERIALS AND METHODS

Eradication operations

The National Park Service (NPS) implemented a management project for squirrels throughout Biscayne National Park in September 2007 through trapping and humanely eliminating squirrels on National Park islands, follow-up population monitoring, survey, and retreatment. The eradication effort began on Old Rhodes Key and Sands Key, in order to eliminate outlying squirrels that likely originated from Elliott Key. This minimised the risk of squirrels spreading to additional islands and/or to mainland Florida. Efforts were then focused on the main population on Elliott Key.

Biscayne National Park is the largest marine park in the US National Park system, with 95% of its 70,000 hectares covered by water and few terrestrial resource management staff. The squirrel eradication was conducted and coordinated by staff of the NPS Florida and Caribbean Exotic Plant Management Team. Biscayne National Park staff provided project oversight, planning and logistical support, and assistance.

Mexican gray squirrels use cavity nests in addition to constructing dreys. However, there are few trees with cavities on Biscayne National Park islands. Because cavities are a limited resource, nest boxes were an effective attraction as nest sites for squirrels. Squirrels in the nest boxes were then flushed into cage traps and euthanized. Nest boxes were also useful for squirrel population monitoring, with their use by squirrels acting as an indicator of missed individuals during the eradication project. There is anecdotal evidence that squirrels will use nest boxes and multiple nests that they have constructed in trees. In light of this, we simultaneously removed squirrels from nest boxes and physically removed dreys and their inhabitants. This proved to be an effective and humane method for removing the entire population from the Biscayne National Park islands.

Aerial and ground surveys of the mixed-hardwood forest were conducted following eradication operations to locate any remaining dreys in the canopy. Host trees were flagged and their coordinates recorded using a Global Positioning System to facilitate relocation. Aerial surveys were conducted by NPS staff in helicopters timed with seasonal tropical hardwood hammock defoliation (typically in the spring). Ground surveys are conducted by NPS staff with emphasis on previously identified drey locations.

During eradication, trained personnel returned at sundown to any trees with dreys flagged during the day. Each drey was destroyed and its occupants euthanized using 12-guage shotguns with non-lead ammunition at a safe, close distance. Weapons were fired into dreys from directions that ensured areas utilised by visitors (such as marina, buildings, campground) were not in the line of fire. Firearm use by non law enforcement NPS personnel for squirrel management in Biscayne National Park was conducted at the discretion and authorisation of the Park Superintendent in accordance with a specific training syllabus developed for this project. Squirrel carcasses were recovered, stored in freezers, and subsequently sent to wildlife specialists for examination.

Approximately 20 nest boxes were installed near known squirrel populations on Elliott, Old Rhodes, and Sands keys. One nest box was also installed on each of Porgy, Adams, and Totten keys where squirrels had not been observed. These islands are within Biscayne National Park and between the squirrel populations and the Florida Keys to the south. This provided a means of detecting any squirrels migrating towards these keys as well as individuals previously undetected. Nest box construction and placement was conducted in accordance with guidelines developed by the Oregon Department of Fish and Wildlife (http://www.dfw.state.or.us/conservationstrategy/naturescaping/squirrel_nesting_box.asp). The boxes were constructed of rough-sawn cedar, installed before shooting operations started, and were attached to trees using plastic tie straps to prevent damage to host trees. Locally obtained leaf litter was used in each nest box to eliminate introduction of non-native species and to stimulate use by the squirrels.

Monitoring

The removal of all squirrel nests and their inhabitants from each island should have eliminated all squirrels. The nest boxes installed following nest removal provide an immediate place for staff to check for any squirrels missed during nest removal. Monthly visual monitoring of the nest boxes commenced in the summer of 2007. To date, no Mexican gray squirrels have been detected. Inspections of the next boxes will continue monthly for one year.

Twelve camera traps have also been placed systematically throughout Elliott Key and Sands Key in trees at bait stations baited with corn and/or sunflower seeds. Bait stations and cameras are monitored at monthly intervals and will remain in place for a year.

RESULTS AND DISCUSSION

From September 2007 to February 2010, 1410 dreys and 33 Mexican gray squirrels (15 males and 18 females) were removed with an average of 43 dreys per squirrel. As drey removal progressed, the number of dreys and subsequently the number of squirrels declined per unit effort. We anticipate that eradication will be completed in 2011 at a total cost of about US$ 80,000 (Table 1).

This project is the first attempt to eradicate a squirrel population in the State of Florida for conservation purposes. We found no examples in the literature where this had been achieved elsewhere for the conservation of native species.

NPS biologists continue to be concerned about the potential ecological effect Mexican gray squirrels on the habitats and listed species found within Biscayne National Park. It is particularly important to keep the species from reaching the mainland of Florida and the United States.

Table 1 Costs of the Mexican Gray squirrel eradication from Biscayne National Park for the period: September 2007- February 2010

Action	Cost US$
Initial Population Research/Assessment	$18,187
Nest/inhabitant Removal	$27,688
Monitoring	$12,090
Transportation	$6100
Equipment/Supplies	$6200
Total	$70,725

The cryptic daytime habits of the squirrels, their ability to move rapidly through the canopy, and efficiency in building dreys meant that constant pressure was required to achieve eradication. Additional funding and staff time are still required to ensure that eradication is completed. However, given progress so far, we are now confident that the techniques used to eradicate the Mexican gray squirrels from Biscayne National Park will be successful and cost effective.

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Small Indian mongoose – management and eradication using DOC 250 kill traps, first lessons from Hawaii

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Abstract Human introduction of the small Indian mongoose (*Herpestes auropunctatus*) has had a catastrophic impact on native fauna of many islands around the world. In Hawaii, the most common method of mongoose control is by using live-traps, followed by euthanasia either by shooting or carbon dioxide poisoning after capture. This is a labour-intensive process, especially as live-traps must be checked every day to comply with humane requirements. The DOC 250 trap was trialled on two Hawaiian islands to test its feasibility as a humane kill-trap for use in mongoose control. The DOC 250 trap was effective in humanely killing mongooses. The DOC 250 trap was also effective in trapping mongooses in a landscape setting. In combination with best-practice wooden trap boxes, the DOC 250 caught more mongooses than live-traps, which were made of wire mesh. The DOC 250 traps should be used in future mongoose control operations in Hawaii as a humane and cost-effective alternative to live-trapping. This trial was a collaborative initiative between the New Zealand Department of Conservation, US Fish and Wildlife Service, Haleiwa (Oahu, Hawaii), US Fish and Wildlife Service, Honolulu (Oahu, Hawaii), US Fish and Wildlife Service, Kealia Pond (Maui, Hawaii) and the Oahu Army Natural Resource Program, Schofield Barracks (Oahu, Hawaii), USA.

Keywords: *Herpestes auropunctatus*, invasive predator, humane control, NAWAC, island conservation

INTRODUCTION

The small Indian mongoose (*Herpestes auropunctatus)* is a catastrophic invasive predator of the West Indies, Hawaiian Islands, South America, Fiji, Mafia Island and island habitats of Africa, Asia and Europe (Long 2003; Warren and Conant 2007). They impact upon and cause the extinction of many species of birds, mammals, and insects (Warren and Conant 2007).

To date in Hawaii, the most common control method for mongoose is cage live trapping. This method requires skilled and intensive labour as traps must be checked daily and captured animals either dispatched on site with firearms or offsite in carbon dioxide chambers. These labour intensive methods impact upon management decisions particularly regarding the size and scope of control projects. Less labour reliant and more cost effective tools are needed to enable control or eradication of mongooses over larger areas, such as on islands and in large mainland reserves.

The DOC 250 kill trap has passed National Animal Welfare Advisory Committee (NAWAC) humane guidelines for use against mustelids in New Zealand (Poutu and Warburton 2005). These traps are always set in trap boxes and are triggered by the weight of an animal stepping onto a treadle. It has been developed for use with four pest species in New Zealand, including the ferret *(Mustela furo)*, which is comparable in size and behaviour to the mongoose.

Given similarities between small Indian mongooses and ferrets, DOC 250 traps should show equivalent humane efficacy for both species. A preliminary controlled test was therefore organised in 2007, to determine whether the DOC 250 kill trap could conform to NAWAC requirements and render small Indian mongoose irreversibly unconscious within three minutes of being caught.

This paper presents the outcome of the preliminary humane test and results from three subsequent, small-scale field trials that used DOC 250 kill traps to target and successfully kill small Indian mongoose in Hawaii.

METHODS

A collaborative programme to test the DOC 250 traps was set up between the New Zealand Department of Conservation, US Fish and Wildlife Service, Haleiwa (Oahu, Hawaii), US Fish and Wildlife Service, Honolulu (Oahu, Hawaii), US Fish and Wildlife Service, Kealia Pond (Maui, Hawaii) and the Oahu Army Natural Resource Program, Schofield Barracks (Oahu, Hawaii), USA.

DOC 250 kill traps were set in current best practice wooden trap boxes (Fig. 1) in the initial humane test and at each of the three sites where field trials were undertaken.

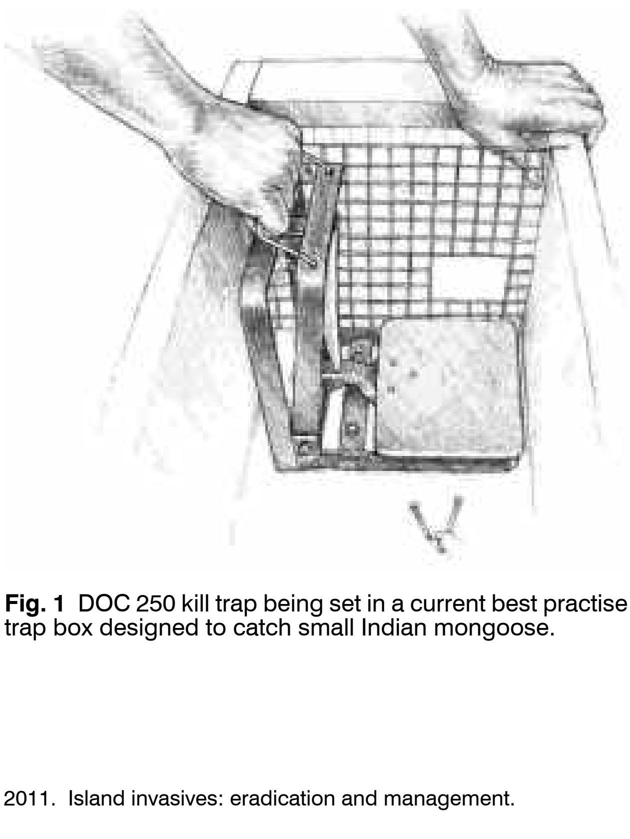

Fig. 1 DOC 250 kill trap being set in a current best practise trap box designed to catch small Indian mongoose.

Pages 225-227 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

Fig. 2 DOC 250 kill trap baited with sardines. The small Indian mongoose has been humanely killed.

Each trap box was baited with tinned sardine or cat crunchies (commercially available cat food in the form of cereal based pellets) and all traps were checked and serviced after between one and three days at all sites.

The objectives of this trial were to determine whether DOC 250 traps are: 1) capable of humanely killing small Indian mongoose; 2) capable og trapping mongooses in the wild; and 3) more effective than Tomahawk live-traps in controlling mongoose numbers in the wild. NAWAC guidelines were used as the humane standards for this trial, as there was a clear mandate to do so from the two governments involved.

Preliminary Humane Test: James Campbell/Ki'i Wildlife Refuge, Oahu Island.

Two captive small Indian mongoose were trapped under controlled conditions in a DOC 250 kill trap set in the marsh/ introduced grass area of James Campbell/Ki'i Wildlife Refuge on 26 March 2007. Both animals were immediately rendered irreversibly unconscious. This was determined by measuring the palpebral reflex, and time to heart-stop, from the moment of trapping. The cause of death in both cases was multiple skull fractures (Fig. 2). This result confirmed the hypothesis that DOC 250 kill traps would humanely dispatch Small Indian mongoose within the NAWAC guidelines and provided the confidence for field trials to proceed.

Table 1 Results from Trial 1, DOC 250 kill traps, on the Jeffrey property, Hilo

Mongoose	Date	Sex	Age class
1	02/04/07	Male	Adult
2	02/04/07	Female	Adult
3	02/04/07	Female	Adult
4	02/04/07	Female	Adult
5	02/04/07	unknown	unknown
8	03/04/07	Male	Adult
9	03/04/07	Male	Adult
10	03/04/07	Female	Adult
11	03/04/07	Male	Adult
12	04/04/07	Male	Adult

Table 2 Results from Trial 2, DOC 250 kill traps

Date	Traps checked	Captures
Sat 16 May 2009	yes	1
Sun 17 May 2009	no	
Mon 18 May 2009	yes	6
Total		7

Trial 1: Jack Jeffrey's property – Hilo, Hawaii Island

Six Doc 250 trap sets were placed 20 - 50 metres apart in rough grassland surrounded by wooded farmland and tree plantings. Each trap was baited with tinned sardines and activated for two nights from 02 - 04 April 2007.

Trial 2: James Campbell/Ki'i Wildlife Refuge – Oahu Island

Fourteen DOC 250 trap sets were spaced 50 - 70 metres apart along access ways within the 45 hectare refuge, composed of wetland with introduced grasses. Traps were baited with tinned sardines on 16 - 18 May and checked twice during the three night trapping period.

Trial 3: Kealia Pond National Wildlife Refuge – Maui Island

Twelve trap sites, comprising a paired arrangement of a standard Doc 250 trap-set placed 1 - 3 metres from a Tomahawk live cage trap, were established at 20 – 50 metre intervals, over an area of marshland and introduced grasses. All traps were baited with cat crunchies placed in a bait jar with wire mesh lid. This paired trap trial ran for a period of four months from 14 June to 17 October 2009, with trap checks and servicing being undertaken every day. In this trial, the Tomahawk traps were checked every day, in accordance with NAWAC humane guidelines. The DOC 250 traps may be checked less often, as these humane kill-traps comply with NAWAC guidelines, regardless of time between trap-checks.

Table 3 Results from Trial 3, DOC 250 and Tomahawk cage trap, paired trial.

Session	Date	DOC 250	Cage trap
1	14 – 20 June 2009	0	0
2	21 – 27 June 2009	0	0
3	28 June – 4 July 2009	1	1
4	5 – 11 July 2009	4	0
5	12 – 18 July 2009	0	1
6	19 – 25 July 2009	0	2
7	26 July – 1 Aug 2009	0	0
8	2 – 8 Aug 2009	3	0
9	9 -15 Aug 2009	0	0
10	16 – 22 Aug 2009	1	0
11	23 – 29 Aug 2009	0	0
12	30 Aug – 5 Sept 2009	0	0
13	6 – 12 Sept 2009	1	0
14	13 – 19 Sept 2009	0	0
15	20 – 26 Sept 2009	3	0
16	27 Sept – 3 Oct 2009	1	0
17	4 – 10 Oct 2009	0	0
18	11 – 17 Oct 2009	0	0
Total		15	4

RESULTS

Trial 1: Jack Jeffrey's property – Hilo, Hawaii Island

One trap malfunctioned and has been discounted. Results are based on five operative traps set for two nights. Five mongooses were caught each night, i.e. ten mongooses in total (Table 1). Each mongoose was killed through extensive skull fractures in the same efficient manner as the two mongooses used in the preliminary humane test..

Trial 2: James Campbell/Ki'i Wildlife Refuge – Oahu Island

A total of seven mongooses were caught in fourteen traps set for four nights (Table 2). All were killed by skull fracture injuries as previously described.

Trial 3: Kealia Pond National Wildlife Refuge – Maui Island

Nineteen mongooses were caught during this trial; fifteen in DOC 250 kill traps and four in Tomahawk cage traps (Table 3).

DISCUSSION

The preliminary humane test and three field trials showed conclusively that DOC 250 kill traps, secured and set correctly in current best practice wooden boxes, are extremely effective at catching and humanely killing small Indian mongoose.

It is interesting to note that mongooses were much more inclined to push through two small, offset apertures and get caught in a DOC 250 trap set in the close confines of an enclosed wooden box, than to freely enter the wide open door of a Tomahawk cage trap. This may be due to the similarity between enclosed trap boxes and natural burrows and crevices, which are natural dwellings for small Indian mongooses.

DOC 250 traps are lightweight and cost-effective, with potential to effectively manage mongoose populations in Hawaii. Best-practice methods for their use have been well developed in New Zealand. These procedures include a formalised maintenance schedule when using DOC 250 traps, to ensure that they continue to perform and comply with humane requirements (DOC Ferret control – kill trapping current best practice guidelines, 2005).

Effective kill traps do not require daily checking, an advantage which allows managers to better utilise labour and maximise cost effectiveness. Early indications from this project suggest that use of DOC 250 traps will enable control and/or eradication of mongoose when applied in "landscape style" trapping operations (e.g., 1000 ha – 21,000 ha), similar to successful projects in New Zealand such as the Whenuakite Kiwi Care Project (Coromandel) and the Resolution Island Stoat Eradication Project (Fiordland). Both projects are based on proven, current, best practice methodologies (Brown 2003; McMurtrie *et. al.* 2008).

Potential gains for conservation that have been made through the trials in Hawaii are a direct consequence of collaboration between several Government agencies in New Zealand and Hawaii and the cooperation and assistance of local landowners. The pooling of technical expertise combined with local knowledge of target and non-target species, local conditions and community requirements enabled effective project planning, and provision of practical support in undertaking the field trials.

CONCLUSION

This study confirms that DOC 250 traps in protective boxes provide a new and more efficient tool for the management of small Indian mongoose than current methods utilizing cage traps. This, however, is merely a start. Formal independent humane accreditation should now be sought to add mongoose to the list of DOC 250 approved target species. More comprehensive research and testing should also be undertaken to ascertain the most appropriate strategies to apply when deploying this equipment in the field. Additional collaborations, such as those undertaken during this study, would be a positive way to achieve these goals.

ACKNOWLEDGEMENTS

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Planning processes for eradication of multiple pest species on Macquarie Island – an Australian case study

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Abstract Eradications of multiple target species challenge managers, especially those seeking to conduct simultaneous eradication programmes. Macquarie Island presents additional challenges because of its remoteness, large size, terrain, weather, and mix of target species. Long lead times for planning are required, reflecting the scale and complexity of logistics and regulatory requirements. Many components of eradications are contingent on initial feasibility and planning decisions. The best options for eradicating target species must be selected early in the planning timeframe. Planning for the eradication of ship rats (*Rattus rattus*), house mice (*Mus musculus*), and European rabbits (*Oryctolagus cuniculus*) on Macquarie Island used many concepts and techniques from previous eradications. These experiences identified that both species of rodents would take the same baits, that the only feasible distribution method was aerial application, and that mice could be harder to eradicate than rats. Other campaigns indicated that most rabbits would consume bait, but some would not, meaning that rabbits were unlikely to be eradicated by aerial baiting. Comprehensive follow-up hunting would therefore be required to remove surviving rabbits, with their detection best assisted by trained detector dogs. These factors formed the basis for bait trials, the results of which were used in a permit application to use the toxin (unregistered in Australia for rabbits) and in state and federal environmental impact assessments. Since approvals are specific to the toxin, techniques and/or bait nominated in applications, commitment to the selected method increases as planning evolves. Procurement priorities were also determined by these early decisions. Dog training was expected to take two years and was the first major procurement item. Bait, bait pods, shipping and helicopter contracts were also required, some of which were interlinked. Intended rabbit eradication techniques also determined staffing levels and the equipment required to support them. Approximately half of the projected costs are associated with post-baiting rabbit hunting.

Keywords: Planning, logistics, aerial baiting, regulatory environment, dog training, rabbit, *Oryctogalus cuniculus*, ship rat, *Rattus rattus,* house mouse, *Mus musculus.*

INTRODUCTION

Macquarie Island (12,780 hectares) is a World Heritage site administered as part of the Australian state of Tasmania. The island is in the Southern Ocean (54°37'53"S, 158°52'15"E) approximately 1500 km from Tasmania and 1000 km from Bluff, New Zealand (Fig. 1). Early European activity centred on commercial exploitation of seals and later penguins, and continued until 1919. Sealing and oiling gangs deliberately or inadvertently introduced numerous alien species. Some species, such as dogs (*Canis familiaris*), established wild populations that subsequently died out. Others, including sheep (*Ovis aries*) and goats (*Capra hircus*) were maintained for domestic use. Five species established feral populations with significant detrimental effects on native flora, fauna and landscapes: ship rats (*Rattus rattus)*, cats (*Felis catus)*, house mouse (*Mus musculus)*, European rabbits (*Oryctolagus cuniculus)* and weka (*Gallirallus australis scotti)*.

Rabbits were introduced to the island for food in about 1879 (Cumpston 1968). Grazing impacts were observed in the 1950s (Jenkin 1975; Taylor 1955) and by the 1960s there was increasing concern about damage to vegetation (Costin and Moore 1960). Rabbit control commenced with the release of the myxoma virus in December 1978, with annual releases until 2006 (although stocks used in the last few years had an expiry date of 2002). Initial control of the population was achieved within five years as myxomatosis spread through the population (Brothers and Copson 1988) but was reduced in its effectiveness after 20 years (Dowding *et. al.* 2009).

Rodents were recorded from the early 20th century, although mice may have established before 1830. The rodents probably established from shipwrecks or were landed with stores (Cumpston 1968). The impacts of rodents are less visible, but damage includes suppression of invertebrate and seabird populations, especially burrowing Procellariformes, and impeded plant recruitment and flowering (Shaw *et. al.* 2005).

Cats were introduced shortly after the island's discovery in 1810, and co-existed with two species of endemic land bird until the establishment of rabbits allowed their population to expand (Taylor 1979). Both land bird species, a parakeet and a rail, were extinct by 1895 (Taylor 1979). Before the introduction of myxoma virus, in the mid 1970s cats annually killed about 60,000 seabirds (Jones 1977). Cat control commenced in about 1974 and emphasis shifted to eradication from 1984. An abatement plan prepared in 1996 (Scott 1996) resulted in additional resources from 1998. With increased hunting effort, cats were eradicated by 2001 (Copson and Whinam 2001).

Weka were introduced to Macquarie Island at about the same time as rabbits, also as food. After rabbit numbers were reduced by myxomatosis in the early 1980s, weka came under increasing predation pressure from cats. Weka were eradicated by 1989, primarily by shooting (Copson and Whinam 2001).

Rabbit numbers began to increase in the late 1990s and by 2000 there was increasing concern about grazing damage to vegetation (Parks and Wildlife Service unpublished data). Awareness of rodent impacts was also growing. With the successful eradication of Norway rats on Campbell Island (11,300 ha) (McClelland 2011) plus increasing numbers of rodent eradications worldwide, similar measures were considered for Macquarie Island.

This paper outlines how decisions taken early in the process of planning simultaneous eradication of rabbits and rodents on Macquarie Island, along with some explicit expectations and assumptions, led to an increased commitment to these early decisions as planning progressed. Those commitments then influenced logistical requirements, many of which were sequential in nature and could not be determined until preceding decisions were made. I reinforce the importance of undertaking trials of techniques and materials to inform subsequent planning components, because early decisions and recommendations can increasingly commit planners to the proposed course of action as approvals and permits are secured.

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INITIAL PLANNING FOR RODENT AND RABBIT ERADICATION

The first challenges faced by those planning the eradication operation on Macquarie Island were remoteness, climate, island size and terrain. This combination of challenges meant that any operation attempting to eradicate three species needed commensurate resources in staff, supplies and equipment. This in turn meant that securing funding would also be a significant challenge. Experiences on Campbell Island proved that Norway rats could be successfully eradicated from large sub-Antarctic islands, but the mix of three target species found on Macquarie was uncommon in island eradication projects. The same mix of species was on Saint-Paul Island (800 ha, 38°42"30'S, 77°32"30'E), where a helicopter-based eradication project with follow-up rabbit hunting succeeded in eradicating rabbits and ship rats, but not mice. The survival of mice may have been related to issues with spreader malfunction and bait spoilage (Micol and Jouventin 2002). The remoteness of Macquarie Island challenges deployment of staff and supplies to the island; as there is generally only one resupply voyage annually to the Australian Antarctic Division (AAD) Station (Fig. 1), and up to 10 tourist vessel visits in summer, with limited available berths. The climate is cool, wet, windy, and cloudy – suggesting immediate issues for the condition and longevity of bait pellets and for flyable weather in which to spread them by helicopter. The terrain is mostly traversable by foot but there are sections of cliffs, steep faces and coastlines which cannot safely be traversed by people, increasing challenges to achieving rabbit eradication.

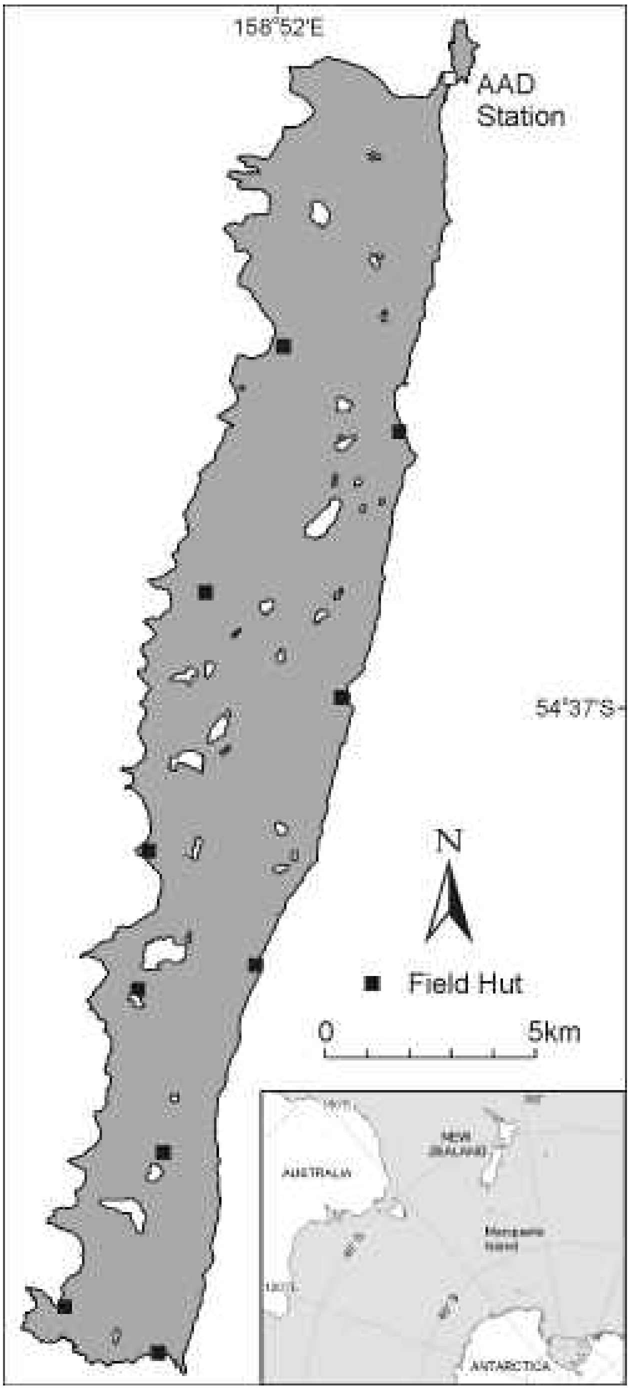

Fig. 1 Macquarie Island showing the location of the Australian Antarctic Division (AAD) Station base station (also known as the ANARE Station) and field huts.

A draft eradication plan was prepared in 2004 (PWS unpublished data), and a project officer was appointed in 2004 to prepare an overview of the situation, recommend eradication methods, and draft an operational plan (PWS 2007). Concurrently, tests of bait weathering and palatability, non-target response to bait, and rodent distribution on the plateau were conducted on Macquarie Island in the autumn and winter of 2005 (PWS 2009). The evaluation of options, the recommendations in the draft eradication plan, and the trial results formed the basis for many of the subsequent planning actions, which commenced in 2007 and continued until the end of the planning phase in May 2010.

Planning lapsed from September 2005 until October 2006, when a further 12-month project officer position was established. Eradication planning was interrupted by the need to prepare a case for funding. The Tasmanian and Australian governments announced joint funding of the project in June 2007, with a projected budget of $24.7 million. From this point, planning could focus on the requirements for eradication. Components of the plan were identified and separate but inter-related plans prepared for the project, comprising 10 parts: A - Eradication Plan Overview; B - Operational Plan; C. - Environmental Impact Assessment; D - Occupational Health and Safety Plan; E - Project Biosecurity Plan; F - Monitoring Plan; G - Communications Plan; H - Project Plan; I - Procurement Plan; J - Staff Recruitment and Training Plan.

Many of the subsequent planning decisions and the sequence of regulatory and procurement processes hinged on key recommendations and assumptions, particularly the choice of toxin and its method of delivery.

Brodifacoum was recommended as the most suitable toxin to attempt rodent and rabbit eradication on Macquarie Island, on the basis of the susceptibility of target species and its documented success in island pest eradications (e.g., Howald *et al*. 2007). Pestoff 20R (Animal Control Products, Wanganui, New Zealand) was selected for the 2005 trials as a suitable bait to carry the toxin, because it had been used on Campbell Island (after testing of various bait types in 1999), and proven success in other island eradications (http://www.pestoff.co.nz/start.htm). Brodifacoum can eradicate rats and mice, although reasons for previous mouse eradication failures where mice co-existed with rats are unclear (MacKay *et al*. 2007). Aerial broadcast was recommended as the only feasible delivery mechanism on Macquarie Island.

Rabbits are also susceptible to brodifacoum (Crosbie *et. al.* 1986; Godfrey and Lyman 1980; Godfrey *et. al.* 1981). Not all rabbits consume bait (Torr 2002), but kill rates >95% are likely. Given the size and terrain of Macquarie Island and a rabbit population estimated at about 124,000 in 2006 (Terauds 2009), comprehensive follow-up ground hunting would be necessary to mop up survivors.

Implications of key aspects

The 2006 Eradication Plan (PWS 2007) determined the regulatory, procurement, planning and budgeting processes for the project, which commenced after appointment of a project manager in August 2007. It soon became apparent that aerial baiting could not begin for at least three years, i.e. winter 2010. Two bait drops were planned, with the second drop targeting rabbits in high density areas and rodents that may not have had access to bait during the first drop.

Regulatory implications

Brodifacoum is not registered in Australia either for aerial application or for use against rabbits. A permit was thus required from the Australian Pesticides and Veterinary Medicines Authority (APVMA). Applications to this agency were known to take a considerable time for assessment, so preparing and submitting an application was a priority. The APVMA has a particular interest in impacts on human health and on non-target species. The New Zealand Department of Conservation (DoC) review on brodifacoum (Fisher and Fairweather 2006) was invaluable for both of these aspects. Reports on non-target species trials undertaken on Macquarie Island added essential information specific to the treatment area, as the project involved species not commonly found on mainland Australia. An application for a Minor Use Permit was lodged with APVMA in June 2008, and the permit issued in May 2009. The use of a consultant to prepare the detailed information in the required format was vital to having the application assessed without further information being sought by APVMA, which would have extended the timeframe still further.

Brodifacoum is not an approved pesticide for use against rabbits in Tasmania; hence an application was also made to the Animal Welfare Advisory Committee and it was recommended for use on Macquarie Island under the state *Animal Welfare Act 1993*.

The scale of the project and the island's World Heritage status required referral of the eradication project to the Commonwealth Department of Environment, Water, Heritage and the Arts under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act). The Environment Minister could then determine whether the proposed project was a controlled action and whether conditions should be imposed on its implementation. An Environmental Impact Statement (EIS) was prepared, which incorporated the 2005-6 bait and non-target species trials, and subsequent trials undertaken in 2007 and 2008 to further assess non-target species impacts. These latter included the results of over-flights of king penguin (*Aptenodytes patagonicus)* colonies. These trials provided information for the assessment of environmental impacts that was not available in the published literature or in Australian operational experience. The EIS acted as supporting information for the EPBC Act referral and included actions to minimise impacts on non-target species and the environment within which baits would be spread. Following a public notification period, the Minister determined in October 2009 that the project was not a controlled action as long as it was undertaken in the manner specified in the referral.

In addition, the state environmental impact assessment process needed to be completed. The required Reserve Activity Assessment was approved in July 2009.

Procurement implications

Trials on Macquarie Island using non-toxic Pestoff 20R bait confirmed its suitability. The bait to be used contains brodifacoum at a concentration of 20 parts per million. State Treasury Instructions require all purchases over $100,000 to be undertaken by public tender. However, a public tender would not have delivered alternate bait with the proven track record of Pestoff 20R (especially in sub-Antarctic conditions). In addition, all trials undertaken on Macquarie Island over three years, the preparation of APVMA permit applications, and the EIS would be nullified if a different bait was selected; weathering and non-target palatability characteristics may not apply to different bait formulations. An exemption from the requirement to tender was thus sought, which effectively required preferred supplier status for Animal Control Products to enable procurement to proceed. Tender processes can take from six weeks to several months to complete, so the exemption received from Treasury in late 2008 allowed the project team to arrange bait orders with greater certainty, within a shorter time frame, and maintain project timelines.

In addition to processes driven by initial project decisions, the results of one tender sometimes influenced the specifications required for the next. For example, the helicopter tender determined the number and type of helicopters to undertake the aerial baiting. Only once the helicopter model (and thus lifting capacity) was known could a tender be let for bait pods (containers) used to transport and store the bait. Once the number of helicopters and the quantity of bait and bait pods was known, a tender was let for a vessel to support the project by delivering passengers, helicopters, bait and fuel to Macquarie Island and to retrieve helicopters at the conclusion of the aerial baiting phase.

Planning implications

Given that rabbits had never been eradicated from large islands with toxic baits alone, previous rabbit eradication operations (Torr 2002; Micol and Jouventin 2002) were analysed and two key requirements emerged.

Use of dogs

The first requirement was that highly trained dogs must be used to detect surviving rabbits, particularly as vegetation recovered from grazing. Dogs trained to the standards required cannot be acquired as an 'off the shelf' item and especially not in the numbers required. Procurement decisions early in the project timeline thus focused on dogs. The timeline allowed for up to two years to train dogs to effectively locate rabbits, to ignore non-target species, and to display absolute obedience to the handler. Timing of the deployment of fully trained dogs, therefore, had to synchronise closely with the date of the intended bait drop.

Handlers can be particularly effective when working with their own trained dogs. However, this model posed an insurmountable risk. With a minimum of six dogs required each year for up to five years, at least 30 highly trained dogs would have been required at specific timeframes for 12-month deployments. The likelihood of finding six people each year with the requisite hunting and dog-handling skills, same standard of dog training, and the personal qualities to work harmoniously in a small isolated community, was considered remote. Furthermore, if suitable handlers (with trained dogs) available for a 12 month deployment could not be recruited each year in time for the ship's departure, then detection dog capacity would drop, pressure on surviving rabbits would ease, and there would be an increased risk of eradication failure. There was also a risk with variability of training standards that the extensive wildlife present on the island may be susceptible to disturbance by dogs that were not properly controlled.

Consequently, an alternative model was developed with trained dogs procured by Tasmania Parks and Wildlife Service (PWS) following a tender process, the dogs remaining the property of PWS, and remaining on Macquarie Island throughout the project. The training standard for the dogs was developed by PWS and adapted from the DoC predator dog and protected species dog programmes, based on 'initial' and 'final' certification levels. The training standards incorporated Macquarie Island-specific aspects, recognising the need to avoid disturbance to non-target species (including dense penguin colonies and extensive seal populations) and an ability to work in steep terrain.

The training of 12 dogs for PWS was spread across three contractors to minimise risks of non-delivery. Additional dogs were trained as backup for any dogs that failed their final certification assessment. Deploying 12 dogs also allowed for mortality of up to 50% of dogs during the course of the project without dropping below the desired minimum of six (which was based on the number of hunting blocks on Macquarie Island). A dog training coordinator was employed to oversee the consistency of training between contractors (two in New Zealand and one in Australia) and conduct the necessary certifications. This process managed the risks of an inconsistent supply of dogs over the course of the project. The converse risk was that by employing new dog handlers to work on Macquarie Island each year the dogs needed to adapt to different handlers annually (or more frequently) throughout the project. This approach could reduce the effectiveness of a well-established hunter-dog team, and required PWS dog handlers to be instructed in how the dogs had been trained to work. However, this seemed to be a lower risk to project success than the risks of not sourcing suitably trained dogs, or dogs with variable training standards. The decision to procure dogs for the duration of the project helped determine the breeds of dogs to be used. Because they had to have a strong hunting drive and be amenable to working for different handlers, Springer spaniels and Labradors were the breeds chosen. The time taken to seek industry advice, prepare and manage tender documentation, draft training standards, prepare contracts for successful tenderers, and allow for pups to be born two years before deployment added another 10 months to the project timeline. As a result, aerial baiting could not begin before the winter of 2010.

Hunting pressure

The second requirement was for sufficient staff to be available after aerial baiting to ensure that the rabbit population continued to decline, rather than breed faster than hunters could kill them. This requirement had significant logistical implications. The only regular voyage for staff deployment is through annual logistics support provided by the Australian Antarctic Division (AAD), who also provide food, clothing and accommodation on the island. Thus, planning for pest eradication on Macquarie Island could not proceed without close liaison with the AAD, and required their commitment to the project goals in order for them to engage with the staff and logistics resources required to support the eradication team in the field.

Support for these field teams also needed assessment of the likely duration of rabbit hunting. In the 2005 draft plan, this was estimated as three years post-baiting, with a further two years of monitoring for sign of rats, mice and rabbits. Hunting teams present over such extended periods also required extensive support in the field. Planning included additional field huts to provide ready access to plateau, west and south coast areas; annual resupply of food, equipment and fuels to new huts as well as to existing huts on the island (maintained by AAD); clothing suitable for extended year-round field work in sub-Antarctic conditions and, crucially, field equipment designed to give hunters the best chance of eradication success. Rabbits are traditionally hunted with firearms, but other methods are used to minimise disturbance to surviving rabbits, and to suit the individual location of rabbits once located. Accordingly, traps, burrow fumigants and nets were purchased, as well as .17 HMR rifles and a small number of .223 rifles and 12 guage shotguns. Additional field equipment included excavating tools, laser range finders, spotlights and filters, night vision and thermal imaging equipment, binoculars, traps, fumigants, smoke generators, GPS units, satellite phones, VHF radios and a range of consumable items and outdoor equipment. The effectiveness of hunters will be enhanced by training in the use of all hunting techniques, as well as in the principles of eradication.

Bait application rate calculations were based on accurate sowing and providing sufficient bait for target species, baits cached by rats or consumed by dominant rabbits before losing their appetite, and the need for sufficient baits remaining to allow access by mice and sub-dominant rabbits. The planned second bait drop is largely to extend the period that bait is available and to ensure that interactions within or between species have not prevented some individuals from encountering bait.

Finally, trials undertaken on Macquarie Island to ascertain the suitability of techniques, materials and equipment have included: 1) aerial distribution of non-toxic bait containing pyranine to determine palatability; 2) weathering of baits; 3) palatability of baits to non-target species; 3) the effects of helicopter over-flights of king penguin colonies; 4) collection of rodent samples for DNA analysis; 5) trials of bait storage pods of different materials and 6) tests of assorted field equipment.

Budget implications

Recommendations and decisions made early in the planning process were crucial in preparing a realistic budget. Trials with Pestoff 20R baits in 2005 enabled reasonably accurate costing of bait and their transport to Macquarie Island. During project planning, budgets were revised to reflect the additional costs of such a challenging project. After an initial estimate of approximately A$12.5M in 2005, the final estimate increased to approximately A$20M. A project contingency of 20% was added to reflect unknown aspects such as fluctuating fuel prices and exchange rates several years ahead of budget preparation, and the dependence of shipping and helicopter costs on weather and fuel. The final budget approved was $A24.7M. Of this, approximately half stemmed from the expectation that rabbits would not be eradicated by aerial baiting, and the long period of post-baiting hunting and monitoring.

One of the most significant early successes of the Macquarie Island Pest Eradication Project was the agreement between the Australian and Tasmanian governments to commit funding to the entire multi-year duration. At that time, the project was expected to take at least eight years. The ability to plan several years ahead without the uncertainty of annual funding applications was a major commitment by government and boosted planning certainty.

CONCLUSION

Early decisions taken in planning any eradication of multiple species need to be based on each species' eradication history. In addition, understanding the characteristics of target species in isolation, and collectively with other target and non-target species is also important. With larger and more complex island eradications, funding can become increasingly difficult to secure. It is vital to recognise that the implications of early decisions can increasingly commit the project to those decisions as the planning process continues. For Macquarie Island, trials of Pestoff 20R baits for weathering, target and non-target palatability informed regulatory permits and approvals. However, had the proposed bait type been changed because of constraints on procurement (tender processes) the trials would have been negated, the environmental impact assessments partly invalidated, and applications to use alternative baits delayed until new trials were conducted. In addition, it would have negated the historical success of the bait in this type of operation. Preparing for the survival of some rabbits after an aerial baiting operation is vital. It is better to have prepared and budgeted for extended follow-up hunting and not need it, than to assume eradication will be achieved by aerial baiting and to find that more work is needed when the budget has been expended. It is usually easier to return any surplus funds than to seek more at short notice.

Some key lessons that apply in particular to large or complex island pest eradications include:

Secure funding commitments for the full project timeframe if at all possible. Not only does this overcome annual funding bids, which if unsuccessful derail the project, it also promotes awareness and buy-in from project sponsors.

Rabbits and mice can be difficult to eradicate, even from small islands. Planning should include comprehensive measures designed to minimise the risk of eradication failure.

Procurement and recruitment by government agencies can be very time consuming and may not reflect eradication staffing or supply needs. Sufficient time and staff need to be allowed for to allow compliance with these processes.

Time and staff resources to complete all of the planning requirements should not be underestimated. This need for planning should be allowed for in project budgets and timeframes to ensure that the operational phase has realistic timeframes allocated to all permit and procurement aspects. Pre-departure workloads are high, especially for ship-based eradications, so additional staff may be necessary to take some of the workload for this busy period.

Peer support is very important and the global network of eradication practitioners readily provides invaluable information, support and experience.

Island-specific trials and comparable island eradication projects provide a sound basis for planning documents. If at all possible, undertake relevant trials on the subject island on target and non-target responses to proposed eradication techniques.

Many procurement aspects are interlinked. Dependencies should be mapped to clarify the critical order of procuring goods and services. For example the amount of bait required needs to be determined to enable the scale of transport to be arranged. If this means chartering a ship the cost implications can be significant.

The basic principles of successful eradication should be at the forefront of planning and are outlined in numerous sources including Broome *et al.* (2005).

Biosecurity (minimising risk of reinvasion) is crucial, may need commitment from non-aligned agencies, and should be planned for and developed from an early stage.

ACKNOWLEDGEMENTS

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The Ka Mate reverse-bait snap trap – a promising new development

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Abstract Development, field trials and potential of Ka Mate reverse-bait snap trap are described. Prototypes were tested on five species of rodents in a range of environments in New Zealand, Alaska, Hawaii, Wake Atoll, Wallis & Futuna Islands, New Caledonia and Seychelles. Paired testing of reverse-bait traps in close proximity to treadle traps was found to be inappropriate because trap function combined with animal behaviour skewed results. The first factory product, the Ka Mate medium "safeTcatch" trap, the corflute "flatpack" trap station and various wax baits are now under evaluation by professional conservation and science practitioners worldwide. One example is Waiaro Sanctuary (Coromandel, New Zealand) where in one year, using only Ka Mate rat traps, 75 ha of forest yielded 656 rats, reducing population indices from 100% tracking tunnel rates to 10%. Data indicates that over 95% of rats were trapped with head/neck strikes, and only one bird was caught in Waiaro in circa 90,000 trap nights using Ka Mate traps set unprotected on the forest floor.

Keywords: Ka Mate traps, reverse-bait snap trap, treadle trap, Victor, Catchmaster, Ezeset, *Mus*, *Rattus*, Wake Atoll, Wallis and Futuna Island, Waiaro Sanctuary, New Zealand.

INTRODUCTION

Advances worldwide in rodent control or eradication on islands during the past three decades have centred on the use of rodenticides (Howald *et al*. 2007). However, the propensity for rodents to develop a tolerance for toxicants (Bailey and Eason 2000) and increasing public opposition to use of poisons may limit their continued use, particularly in mainland situations (Williams 1994; Towns and Broome 2003; Mason and Littin 2003; Towns *et al*. 2006). Traps have similarly evolved in design and strategic use but they also attract a public opposition, ostensibly over animal welfare issues.

The New Zealand Department of Conservation (DOC) requires a better performing snap trap that gives more consistent catch/kill rates; improved animal welfare outcomes; less non-target catch and environmental interference; enable higher quality trapping data; have greater durability; less maintenance; quicker servicing during routine checks; and are easier for operators to use than current preferred rodent traps. In short, better returns from traps in relation to money expended (Keith Broome pers. comm., April 2004).

In this paper, we describe the development and field trials of Ka Mate (KMT) reverse-bait snap traps, which have been designed to meet modern efficacy and animal welfare requirements.

Traditional, wooden based "break-back" traps (snap traps), have been used in New Zealand since at least 1920, particularly for bio-security at ports, rodent control around factories, and as a health measure in urban environments (Wodzicki 1950). They have also been used internationally for scientific data collection and in conservation management programmes (Bull 1946; Watson 1956; Wodzicki 1969; Daniel 1973; Innes *et al*. 1995; Dunlevy *et al*. 2000; Efford *et. al*. 2006; Malcolm *et al*. 2008; Theuerkauf *et al*. 2010). More recently, snap traps have been employed in many large-scale New Zealand mainland island rodent control programmes (Saunders 2000, 2003; Speedy *et al*. 2007; Ogden and Gilbert 2008) and as adjuncts to toxicants in island eradication campaigns (Morrell *et al*. 1991; Taylor *et al*. 2000; Merton *et al*. 2002; Thomas and Taylor 2002; MacKay and Russell 2005; Nugent *et al*. 2007; Witmer and Burke 2007; Varnham 2010).

Rats have been eradicated from at least two islands of up to 21 ha with snap traps (Moors 1985; MacKay and Russell 2005; Howald *et al*. 2007), but trapping is usually considered to be too labour intensive and expensive as a sole eradication technique for rats (Keith Broome pers. comm.). Poor trap performance has exacerbated negative public attitudes, resulting in stricter rules for trapping and animal welfare now embedded in policy and law (Mason and Litten 2003; Powell and Proulx 2003; Litten *et al*. 2004).

Traps have traditionally varied from toggle trigger traps with a small (baited) trigger to large treadle plate designs that use a lure to entice target species to step onto a plate to spring the device. Treadle snap traps are generally easier to use than trigger traps. Many trap designs are operationally unstable and not robust enough to withstand the rigours of long term field use. Baseboards on wooden models warp or split, staples pull and weak points on plastic variations soon break. The larger trigger area of treadle traps makes them more prone to misfire due to environmental events and the presence of non target species.

THE KA MATE REVERSE-BAITING SNAP TRAP

Trap development

During the mid-1980s, two of us (RT and BT) experimented with ways to improve snap trap efficiency. Modifications were made to wooden based trigger "Ezeset" traps being used to catch Norway rats (*Rattus norvegicus*) which led to "reverse-baiting" snap traps with dense, supportive bait beneath rather than on top of the trap trigger. This utilised the bait as a removable structural component of the trap, introducing significantly more stability into the trigger function.

Six steel reverse-bait snap traps were then engineered in 2003 and of the five ship rats (*Rattus rattus*) these first killed, three were cranial and two were humane neck strikes. Fifty of these traps were subsequently incorporated into a 6 month paired trial with "Victor Professional" traps at Weka Bush, Nelson Lakes National Park. In 2005, 100 handmade aluminium prototypes (Fig. 1), which we called Ka Mate (KMT) traps, were integrated with the steel traps into an alternating trap trial with "Victor Professional" rat traps and tested over 13 months in Nelson Lakes Big Bush rodent control area. Another 100 KMT prototypes were included in an alternating trap trial with Victor Professional rat traps in DOC's 2005 trap research programme in Te Urewera National Park.

The KMT traps caught and killed mice (*Mus musculus*), rats (*Rattus rattus*), weasels (*Mustela nivalis*), stoats (*M. erminea*) and hedgehogs (*Erinaceus europaeus*). In the Te Urewera trial, the KMT traps also had far fewer

Pages 233-238 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

Fig. 1 Relative condition of Ka Mate prototype (left) and Victor Professional (right) after equal environmental exposure at adjacent sites in the Big Bush trap trial.

unsprung/bait missing events than Victor traps (2 versus 71, respectively), indicating that the reverse-bait trigger reduced problems with non-target and environmental triggering. When compared with wooden-based wire striker traps, operators also found the aluminium KMT to be the safest to set and handle, easiest to clean and maintain (Fig. 1), required the least service time during routine checks, and had the greatest durability in the field (Paton *et al.* 2007; Morriss *et al*. 2007; Moorcroft *et al.* 2010).

In August 2005 on the Seychelles Islands, Gideon Climo (pers. comm.) undertook three 2 hour evening trapping sessions using six KMT prototype traps, which were systematically set, checked, cleared and rebaited with coconut on a rotational basis. He caught over 60 ship rats, achieving 100% humane head and neck strikes on the adults and predominantly shoulder and mid torso strikes on small rats.

The first Norway rat (*R. norvegicus*) caught in a KMT prototype was on Adak Island, Alaska in May 2006. The technician reported "*a perfect kill just behind the eyes"* and that the unprotected traps remained set and continued to catch after exposure to *"gales whipping vegetation, deluges of rain and burial in snow"* (Lisa Spitler pers. comm.).

On Wake Atoll in October 2007, BT and PD established a 200 x 200 m trapping grid for rats consisting of 100 traps spaced at 20 m. Fifty KMT prototypes formed a central core within the grid and were surrounded by 50 Catchmaster (CM) wooden based trigger traps modified to operate as "treadle" traps. Midway through the trial an extra 32 CM traps were added to the periphery, creating double trap sets on three sides of the grid. All traps were tacked to plywood base boards, placed unprotected on the ground and baited with cubes of fresh coconut. The grid was checked and serviced morning and late afternoon, totalling 13 check periods over 7 days. Wake had a high density rat population and a total of 549 rats (520 *R. exulans* and 29 *R. tanezumi*) were caught – 297 from 650 individual KMT trap checks and 252 from 810 individual CM trap checks. KMT traps scored 157 head/neck strikes to 125 body strikes, whereas CM traps scored 94 head/neck strikes to 152 body strikes and both trap types recorded low numbers of limb and tail strikes. Non-catch interference also varied between trap types, with KMT recording 85 traps sprung/empty and 13 traps set/bait missing, compared to CM with100 traps sprung/empty and 172 traps set/bait missing. Hermit crabs were the only non-targets caught, 6 in KMT and 22 in CM (BT & PD unpublished data). Clearly the KMT traps out-performed the CM traps on Wake, scoring higher catch rates to trap check ratios and a greater percentage of head and neck strikes. The considerable disparity in *trap set/ bait missing* totals is hugely significant, especially since

Fig. 2 Ka Mate "*safeTcatch*" trap - with trigger cowling and wax bait.

it was mechanical malfunction (rectified in seconds with a file) that caused the problem in the small number of KMT traps afflicted whereas learned avoidance behaviour by rats was the cause with the CM traps.

From 2007-2010, KMT prototype traps were used in ecological surveys on New Caledonia, Wallis and Futuna Islands (Theuerkauf *et.al* 2010) and in trials to test the efficacy of unprotected KMT traps against "Ezeset" wooden based trigger traps on Pacific, ship, and Norway rats (Theuerkauf *et.al* 2011). These studies concluded from C. 2900 trap nights that KMT traps were the more effective against rats > 100 g (i.e. predominantly ship and Norway rats), whereas "Ezeset" traps were more effective against rats < 100 g (predominantly Pacific rats). A high percentage of "Ezeset" traps were sprung by heavy rain but rain had no effect on the KMT traps, which maintained a significantly higher percentage of operational traps throughout the trials. The durability of the KMT traps was considered an advantage for long term field use.

The "safeTcatch" rat trap

The first commercial KMT trap to be produced was the "safeTcatch" ("sTc") rat trap (Fig. 2), which incorporates a safe set mechanism and is currently available from KMT Ltd, Nelson, NZ. The traps are constructed from extruded aluminium with stainless steel shafts and fasteners and double sprung with galvanised springs. Bar catches that engage when arming the trap work in conjunction with the wide retainer arm that automatically releases the safety

Fig. 3 Typical humane head strike - ship rat in unprotected Ka Mate "sTc" trap.

catch during setting, which makes the process easier for those with weaker hand strength. The traps are supported with replacement parts, which means that KMT traps can be easily repaired, upgraded or converted as design of component parts develops to improve trigger configuration or to suit a different target species. Replacement of any part can be easily undertaken using a simple custom-designed trap tool, so there is no reason to discard a whole trap.

A detachable plastic trigger cowl forces rats to take the bait from the front of the trap, ensuring a humane head strike (Fig. 3) while reducing potential for learned trap avoidance. Baits held firmly beneath the curve in the trigger ensure the trap will not trigger prematurely when knocked or when non-target species such as lizards, birds, or small mammals walk, crawl, or bounce on the trigger. Since it requires a concerted effort by the rat to remove the bait from beneath the trigger (which can cause the trap to move), it is essential that the trap be restrained for maximum efficiency. Holes are therefore provided in the base for spikes, screws or ties, which enable it to be secured to a backing board or either horizontally or vertically to a natural substrate. The operational stability inherent in the design of the trap (especially the trigger function) reduces spontaneous misfire and by-catch and the simple trap setting procedure minimises operator bias between trappers.

Ka Mate "flat-pack" protective station

Protective covers are used with traps to restrict entry by non-target fauna and to protect the baits, but many covers in use are bulky, heavy, flimsy or difficult to access. Ka Mate has produced a trap station fabricated in one-piece from "Corflute" cellular plastic sheeting. KMT "*flat pack*" stations fold compactly for storage and transport and have a lid that provides easy access (Figs. 4 and 5).

The stations have entrances on each side at one end for rats and centrally placed for mice, which provide alternative avenues for entry or escape and create a 90 degree entry angle that reduces the reach of non-target birds. When stations are fixed with stakes or weighted with rocks on the side flaps, target species can enter and walk up to the trap on natural substrate. Decomposing carcases suppurate directly into the ground. Alternatively, KMT stations and traps can be screwed vertically onto trees/posts/walls at a height that allows target species easy access. When set vertically, the rats are confined to a smaller floor space, preventing pull back as the trap triggers and enhancing catch effectiveness (Fig. 5).

Fig. 4 Corflute "*flatpack*" trap-station (assembled for use and folded for storage).

Bait development

The bait is crucial to the function of Ka Mate traps and requires removal by a positive twist or tug to extract it from beneath the trigger to spring the trap. Rodents invariably take baits by mouth, which ensures the animals' head is in an optimum position to achieve an efficient fatal head strike. Bait can be household food items, such as hazel nuts, brazil nuts, walnuts, dog and cat pellets, chocolate, dried cheese and cubes of fresh coconut, or any other food firm enough to support the downwards pressure of the trigger. KMT has also developed and tested purpose-built baits using "Pestoff" non-toxic pre feed (Animal Products, Wanganui) as a base ingredient. When mixed into palm nut wax with different flavoured additives, the baits can be moulded into plugs of optimum shape and size to fit the KMT trigger (Fig. 2). These baits can be effective for up to a month in dry conditions, but earlier replacement is recommended.

Utilisation and user perception

When the Ka Mate "sTc" trap became available in June 2008, prospective users such as community trapping groups began undertaking trials to test the new traps. They invariably set up proximately paired and/or alternating trap trials with Victor Professional traps and early anecdotal feedback indicated some disappointment over KMT trap performance. The issues apparently arose from long established practices associated with the operation of traditional snap traps, which were problematic when universally applied to Ka Mate traps. For example, trappers assumed that baits placed under the extreme end of the KMT trigger would be easier to remove (i.e. the equivalent of hair triggering old style traps) and consequently improve "sTc" trap performance. The practice instead exacerbated the incidence of rats beating the striker, being injured by a glancing blow or caught by a limb. It also increased the chance of catching non-target species. Furthermore, trappers often did not appreciate that the curved "sTc" trigger that accommodates the bait is specifically designed to slow rats down by forcing them to twist the bait sideways to remove it, ensuring the head is in optimum position to receive an efficient cranial strike.

Fig. 5 Vertically set "*flatpack*" station with the door open.

Similarly, when several users complained that their KMT traps were not achieving high catch rates, it transpired that during service checks any traps found still set were bypassed, with many baits unchanged for two or three months. Contrary to common belief, rodents do not like stale mouldy food and it is imperative that the bait on KMT traps be replaced regularly.

Also, people placed new sterile KMT traps alongside pre-used odour saturated treadle traps, creating an obvious disadvantage for the KMT traps because of rats' inherent nervousness around new equipment. Neophobic behaviour combined with differences in trap function (e.g., the arbitrary depression of the treadle foot-plate vs conscious, controlled reverse-trigger bait removal) tended to skew the trials into a "race" to see which trap would catch the same rat first. Trap catch data and observations made on several occasions indicates that in most instances (unless there is intense competitive pressure) it takes much longer for rats to trigger a KMT trap than a large-plate treadle trap. Rats have been seen to cautiously approach baited KMT traps several times, often from different angles, before even putting a foot on them and they sometimes departed altogether for several hours or overnight before returning to check out a trap again. As their confidence grew, they would on occasion mouth the bait several times or nibble it a little before making the fatal decision to take a firm hold and twist or pull it from beneath the trigger (BT, RT, PD and Gideon Climo pers. obs., Baki Bakhshi video recording). Many trappers fail to understand that the most important function to test for in a new trap is not how quickly it catches rats, but how effectively it kills them.

Since results from several of the field trials raised issues with regard to the validity of proximal paired testing, we considered a well planned, large scale trial was needed to test the efficacy of the commercially produced KMT *"safeTcatch"* rodent traps in isolation of other brands. An opportunity for a major collaborative "*trap trial by management"* arose in late 2008 using "sTc" traps for rat control in a private eco-restoration project in Northern Coromandel.

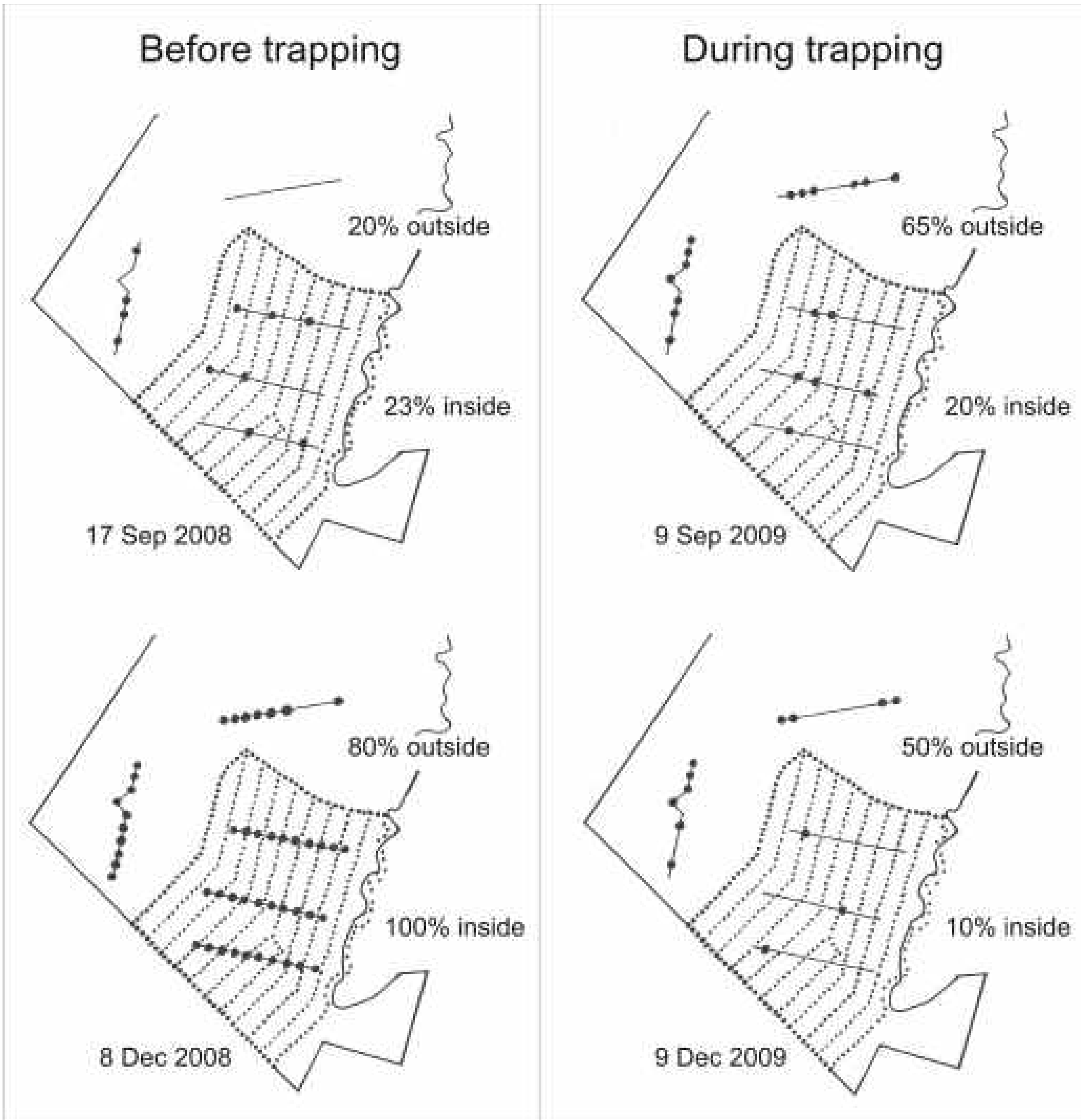

Fig. 6 Waiaro grid layout and Year 1 tracking-tunnel results.

WAIARO SANCTUARY TRAPPING PROGRAMME

Waiaro Sanctuary is private land in the Moehau Kiwi Recovery zone, Moehau Forest, ten kilometres north of Colville. The first phase goal of this new rodent trapping programme was to achieve a toxin-free eradication of ship rats, or to reduce and hold their densities at low levels (5-10% tracking tunnel indices) over a 75 ha block, using only KMT traps.

A 75 ha grid was created with 427 single "sTc" rat traps at 25 m intervals along 10 trap lines spaced 75 m apart, with a perimeter line set along three sides of the grid (Fig 6). Fifty of the perimeter traps were in protective KMT "flatpack" stations. The rest of the traps were fitted with plastic trigger cowls and secured, uncovered, by being pinned to the ground or tied to tree roots. Five index lines (10 tracking tunnels per line) were installed to independently monitor trapping success, three within the grid and two outside (Fig 6).

Traps were given time to weather and three applications of Pestoff RS5 nontoxic pre-feed was hand broadcast sequentially across the block, along trap lines and then in close proximity to the traps only. The traps were then set using KMT RS5 wax plug baits.

Fifteen full checks with all traps serviced in a 24-48 hr period were completed in the 12 months from 22 January 2009 to 21 January 2010, the majority in the first 6 months. Alternatively, progressive servicing was carried out line by line over periods of a week to a month and in winter service checks were restricted to perimeter lines only. Head and neck strikes on adult rats were so consistent that the field teams stopped recording the category, insisting that the KMT traps were achieving "99%" humane kills, including body blows to smaller rats (KM pers. obs.). Four index tracking sessions were undertaken both inside and outside the trapped area - two prior to trapping and two during trapping.

The traps caught 656 rats, with index tracking frequencies reduced from up to 100% before trapping to 10% during the trapping period (Fig 6). An initial knockdown of 299 rats was achieved in less than three weeks with tallies rising to 558 at three months. A further 98 rats were caught during the next nine months with 15 of these in the six weeks before the final January 2010 check (Fig. 7) - a marked contrast to the 117 caught on night one 12 months earlier. The reduction of rats was substantial and only one bird (not identified) and 81 mice were the by-catch from one year's trapping (approx 90,000 trap nights) in Waiaro Sanctuary with unprotected traps.

The RS5 wax plugs remained intact for more than a month, but probably lost their attractiveness as bait for rats much sooner. Operators found the KMT stations convenient to install and access, and the traps easy to operate and service.

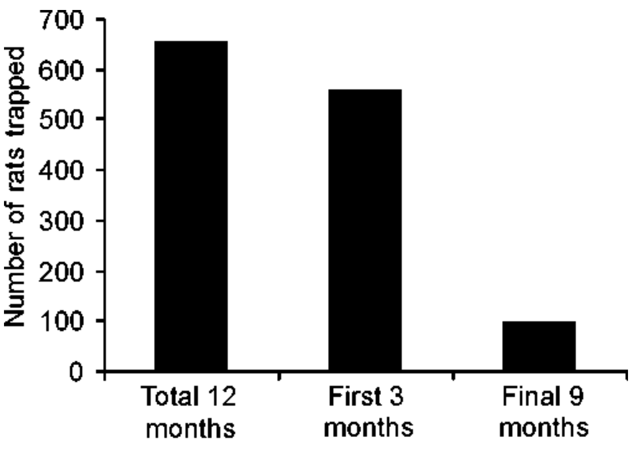

Fig. 7 Waiaro trapping - comparative totals of rats caught by period.

DISCUSSION

Varied outcomes from the early field trials made it difficult to determine what advantages the KMT prototypes provided over the traditional trigger and treadle traps. As the data base grew we began to speculate that rat behaviour coupled with trap function was elemental to the different catch rates being recorded between the trap types, the main contributing factor being that it took rats longer to spring a KMT trap than a traditional trigger or treadle trap. The dense population of Pacific rats on Wake Atoll coupled with the use of night vision equipment (plus the mass of data this project generated) and Gideon Climo's trapping of ship rats in the Seychelles provided the first opportunities to evaluate rat behaviour in conjunction with KMT trap function from direct observation. Although we have drawn our conclusions from all the studies, it is the significant level of rat control achieved with Ka Mate traps in Waiaro Sanctuary that verifies its potential when used alone, unencumbered by the proximity of other trap types (Fig. 7).

The functional stability of the reverse-bait trigger generates a very consistent catch performance. The percentage of quality-kill head and neck strikes can be increased and environmentally generated misfire, rodent induced trap disturbance, and non-target by-catch significantly reduced when using KMT traps. Such results minimise the opportunities for rats to learn trap avoidance and reduces animal welfare issues. The simple standard setting procedure of KMT traps eliminates operator bias and improves population indexing.

The functional stability of KMT traps coupled with trap durability enables traps to be screwed vertically onto bulkheads in ships and permanently fixed inside containers or onto wharves. KMT traps can easily be cleaned and sterilised for bio-security purposes by boiling and are robust enough to operate with minimum maintenance in estuarine and marshland environments. They could be hoisted into trees to sample for rodents in forest canopies or provide protection to hole nesting birds, and are safe enough to be set in many situations where use of other traps would pose a risk to vulnerable non-target species.

In New Zealand, increasing numbers of community groups vie for the same resources to set up predator control programmes and many established projects are continually expanding the areas already being trapped. Development of effective long-life bait will be the key for using traps instead of toxicants to control rats in mainland situations, or for long term surveillance on islands. As trapping technology and deployment improves, wider spacing and less frequent servicing may make it possible to manage larger areas for the same capital outlay, but care must be taken to get the strategy right.

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Rat eradication campaigns on tropical islands: novel challenges and possible solutions

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Abstract Invasive rodent eradications are a proven, effective method of restoring affected ecosystems and preserving biodiversity on islands. Current rodent eradication practices are inherited from successful temperate or subantarctic campaigns, yet a direct translation of this practice to eradications on tropical islands can be problematic. Tropical island rodent eradications face novel challenges not seen in temperate climates, such as land crabs and the use by rats of complex three-dimensional habitats, such as the crowns of coconut palms (*Cocos nucifera*). To enhance the ability to anticipate and respond to such challenges, we studied the role of land crabs in eradication outcomes, and patterns of habitat use (ground *vs.* canopy) by rodents on several tropical islands. During rat eradication planning studies at Palmyra Atoll, Line Islands, indigenous land crabs competed with ship rats (*Rattus rattus*) for broadcast bait pellets. At Dekehtik Island, Pohnpei State, Federated States of Micronesia, rats (*R. exulans*) were frequently observed in the forest canopy and most often in the crowns of coconut palms. A pyranine-based biomarker study conducted at Palmyra found that over 30% of captured rats were not exposed to bait following sequential hand-broadcast applications of 18kg/ha and 9kg/ha. Extensive use of coconut palm canopy habitat by rats and competition for bait with land crabs may reduce rat exposure to rodenticide applied to the ground. The risk of land crab interference and canopy preference to eradication success can be mitigated through increased broadcast application rates and aerial broadcast techniques. Such practices require close scrutiny for potential adverse impacts to non-target species. A better understanding of the eradication challenges inherent to tropical environments may enhance the conservation community's ability to effectively and safely manage the threat that invasive rodents present to tropical island biodiversity.

Keywords: Land crabs, forest canopy, rodenticide efficacy

INTRODUCTION

Palmyra Atoll (250 ha) in the Northern Line Islands, and Dekehtik Island (2.6 ha) off Pohnpei Island in the Caroline Islands, are similar in latitude (5.5°N and 6.5°N, respectively), topography, and forest community structure. Both locations have a history of human use, including species introductions. Palmyra Atoll (Palmyra) is now a National Wildlife Refuge co-managed by the US Fish and Wildlife Service and The Nature Conservancy, and Dekehtik Island (Dekehtik) is managed by the Madolenihm Municipal Government of Pohnpei State in the Federated States of Micronesia.

Both Palmyra and Dekehtik have plant communities typical of low-lying coralline islands and atolls in the tropical Pacific: broad-leaf tree species and coconut palms (*Cocos nucifera*) constitute the forest canopy, and shrubs line the littoral zones (Mueller-Dombois and Fosberg 1998). Monodominant stands of coconut palms represent 45% of Palmyra's forested land area (Wegmann 2009), and 30% of the forest canopy on Dekehtik (Wegmann *et al*. 2007). Both locations have multi-species assemblages of land crabs comprised of Coenobitidae (coconut crabs and hermit crabs) and Gecarcinidae (burrowing land crabs). The estimated mean density of the five species assemblage of crabs at Palmyra is 296 ± 139 (SD) crabs per hectare (Howald *et al*. 2004).

It is unclear when rats (*Rattus sp.*) arrived at Palmyra or Dekehtik. Ship rats (*R. rattus*) likely invaded Palmyra during US military activity in the 1940s (Depkin 2002), whereas Pacific rats (*R. exulans*) likely travelled to Dekehtik with Micronesian voyagers several millennia ago (Matisoo-Smith and Robins 2008). No other invasive mammals were found at either location.

Tropical oceanic islands represent some of Earth's most biologically unique ecosystems, yet the very remoteness that fuels species radiations and high levels of endemicity on islands also renders such systems vulnerable to invasive species. Invasive rodent eradications are a proven, effective method of restoring affected ecosystems and preserving biodiversity on islands (Veitch and Clout 2002; Towns and Broome 2003). Current rodent eradication practices using rodenticides such as brodifacoum are inherited from successful temperate or subantarctic campaigns (Howald *et al*. 2007). However, a direct translation of this practice to eradications on tropical islands can be problematic. Tropical islands present challenges that are not encountered on islands in temperate climates. On tropical islands these include large populations of land crabs and the novel use by rats of complex three-dimensional habitats. Both Palmyra and Dekehtik provided the opportunity to assess the extent to which such problems might influence rat eradication attempts.

Land crabs (crabs) are the main consumers on many tropical islands that lack native mammals (Carlquist 1967; Burggren and McMahon 1988; Green *et al*. 2008). Crabs primarily consume plant material (Burggren and McMahon 1988), such as leaf litter, fruit pulp, and seeds (Garcia-Franco *et al*. 1991; O'Dowd and Lake 1991; Wolcott and O'Connor 1992; Lindquist and Carroll 2004; Green *et al*. 2008). The crabs also opportunistically feed on carrion, or prey on bird chicks (Wilde *et al*. 2004). Crabs readily consume grain-based rodenticide bait pellets and blocks (bait) that have been applied to the ground, and persistently attempt to access bait housed in bait stations (Howald *et al*. 2004; Buckelew *et al*. 2005; Wegmann *et al*. 2008). Crabs also consume the carcases of rats that have consumed lethal amounts of bait (Wegmann *et al*. 2008). Crabs do not appear to be detrimentally affected by exposure to brodifacoum through the consumption of bait (Buckelew *et al*. 2005).

Successful eradications require bait application rates or bait station placement to deliver bait to all rats in every territory for a specified time. Typically this period is three to four days when baits are broadcast, or until all rats are removed when bait stations are used (Wegmann 2008). The success of either approach relies on knowledge of the size of home ranges (or territory), and patterns of habitat use (e.g., tree use *versus* ground use) of the target rat species. For example, if rats spend a significant amount of time in the forest canopy a ground application may not sufficiently expose all rats to rodenticide and additional bait may need to be applied to the canopy. Bait application rates and methods are thus a central issue in the tropics

Pages 239-243 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

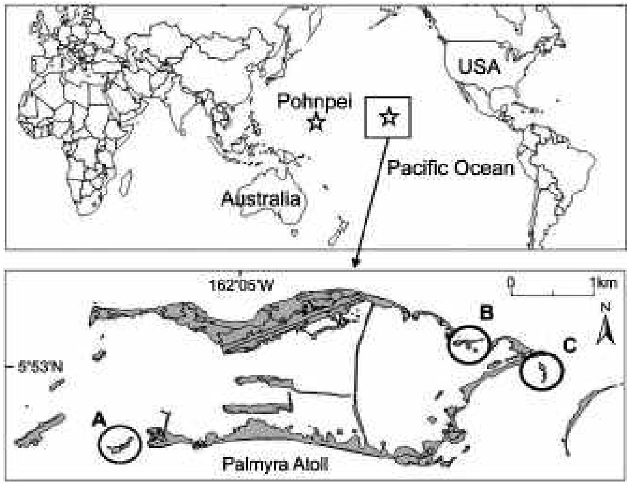

Fig. 1 Location of the field sites : Palmyra Atoll, USA, and Pohnpei, Federated States of Micronesia. For biomarker study sites at Palmyra Atoll - A) Home Island: Area = 1.7ha, distance to adjacent land = 165m, broadcast effort = 33 person-hours. B) Whippoorwill Island: area = 1.9ha, distance to adjacent land = 70m, broadcast effort = 37 person-hours. C) Portsmouth Island: area = 0.8ha, distance to adjacent land = 30m, broadcast effort = 16 person-hours. Distance to adjacent land is the shortest measure from the periphery of the study area to untreated land.

where bait competition by land crabs is high, and canopy habitats are highly attractive to rats. To enhance the ability to anticipate and respond to such challenges, we studied the role of land crabs in eradication outcomes at Palmyra, and patterns of habitat use (ground *vs.* canopy) for *R. exulans* at Dekehtik.

MATERIALS AND METHODS

Crab consumption of broadcast rodenticide bait at Palmyra Atoll

From 19 June to 5 July 2008, 25-W Biomarker bait (Bell Laboratories) was hand-broadcast to the ground and forest canopy on three islands at Palmyra Atoll: Home (1.7 ha), Whippoorwill (1.9 ha), and Portsmouth (0.8 ha) (Fig. 1). Rats were eradicated from Home and Whippoorwill Islands during trial broadcast-based eradications in 2005 (Buckelew *et al.* 2005), but have since reinvaded (Wegmann and Middleton 2008). Before this study, all three islands were surveyed for rat presence and land crab community composition. These islands met the following selection criteria: 1) islands must be isolated by water that is standing throughout the entire tide cycle 2) islands should host land crab and plant communities that are representative of the entire atoll in composition and structure.

Bait uptake was determined using the biomarker pyranine, a hydrophilic, pH-sensitive fluorescent dye. Pyranine is non-toxic, odourless and tasteless, and is fluorescent green when exposed to UV light. While biomarkers are commonly used in wildlife management studies (Fry and Dunbar 2007), the use of pyranine as a biomarker within an eradication scenario is a relatively new innovation (Towns and Broome 2003; Greene and Dilks 2004; Griffiths *et al.* 2008).

Bait was broadcast to the ground and forest canopy at the label-specified maximum application rate (18 kg/ha followed by 9 kg/ha five days later) on Whippoorwill and Portsmouth Islands, and at more than twice the maximum label application rate (36 kg/ha followed by 36 kg/ha five days later) on Home Island. Bait was hand-broadcast by a six-person baiting line in which broadcasters were spaced 5 m apart. The amount of bait consumed by target and non target species was measured in 25 m long x 1 m wide plots on Whippoorwill (18 plots), Home (22 plots), and Portsmouth (7 plots). Wire flags were scatter-planted in each plot, and a single bait pellet was placed at the base of each flag. Plots were calibrated to the different application rates: 9 kg/ha = 10 flags, 18kg/ha = 20 flags and 36 kg/ha = 40 flags.

After the bait broadcast line passed over a bait consumption plot, pellets near the plot were picked up and moved to the flags within the plot. Bait consumption plot sampling began the day after bait application and continued until all baits were removed from each plot. Plots were sampled by checking each flag for the presence of a pellet. Flags without pellets were pulled and tallied. The total number of flags pulled from each plot (total number of pellets removed) was recorded. Bait consumption plot locations were maintained and plots were reset for the second round of bait applications.

Rats were caught in Hagaruma live catch traps that had been seasoned by dipping in cooking oil, placed atop overturned plastic 19 litre (5-gallon) buckets, and baited with solid coconut endosperm smeared with peanut butter. Traps were placed 10 m from each other along transects travelling the length of each island. A total of 1132 trap nights were logged over the course of the study: Home (n = 441.5), Whippoorwill (n = 489); Portsmouth (n = 201.5). Half trap-nights were counted when traps were sprung without a capture. Traps were opened on all islands for two nights after the first bait application, and for four nights after the second bait application.

Captured rats were first inspected for external biomarker sign using handheld UV flashlights. If a rat showed external biomarker sign (paws, anus, tail, mouth), it was euthanased and inspected for internal biomarker sign. Rats without external sign were marked with a permanent marker on the proximal-dorsal section of their tails and released at their point of capture.

Land crab index of abundance (IOA) sampling was on 3 and 4 July between 07:00 and 10:00. Twenty crab IOA transects were surveyed on Whippoorwill and Home, and 10 transects were surveyed on Portsmouth. Transect start points and bearings were randomly selected. The 5 m transect width was maintained by a 2.5 m pole swung in a 180° arc pivoted from the centre of the transect. All crabs seen without searching through the underbrush, beyond the entrance to burrows, or under the leaf litter within a distance of 2.5 m from the central transect axis were counted and identified to species: *Coenobita brevimanus, Coenobita perlatus*, *Cardisoma carnifex*, *Cardisoma rotundum*, and *Birgus latro*. The total number of crabs observed divided by the total area surveyed became the crab IOA value for each study site.

Hermit crabs were collected from each study site every day for eleven days following the initial bait application. Ten hermit crab collection plots (PVC piping driven into the ground) were established on Home and Whippoorwill, and seven were established on Portsmouth. Each sample day, the crab closest to each plot pole was collected and assessed for biomarker presence. Internal biomarker sign was assessed by chilling the crabs in a refrigerator until they were relaxed enough to be safely removed from their gastropod shells. Because hermit crabs are long-lived and important components of Palmyra's terrestrial ecosystem, care was taken to not injure them during the sampling process. All portions of the body were examined for the presence or absence of biomarker sign; biomarker in the crabs' alimentary tracks was visible through the skin of the abdomen. After assessment, hermit crabs were returned to their shells. In some instances, bodies would not come free from the shell easily and such crabs were not assessed.

Crabs were released back to their collection sites on the morning following the day of capture. Of the 330 crabs sampled, four died during captivity.

Habitat use by rats on Dekehtik Island

On Dekehtik Island, habitat use behaviour of seven *R. exulans* was monitored at four-hour intervals for four consecutive days. Rats were captured using Hagaruma live traps and fitted with 4.2g radio-collars (Advanced Telemetry Systems). Digital receivers fitted with directional Yagi antennas were used to track the collared rats. Each radio-collared rat was located every four hours throughout each sample day. It was assumed that the four-hour spacing between samples was more than sufficient time for the rats to recover from any study related disturbance. At each observation, we recorded time, location, whether or not the rat was observed on the ground or in the forest canopy, and if in the canopy the tree species used.

Statistical analyses

We used one-way ANOVA test, with Tukey's pairwise comparisons and assumed significance if $p \leq 0.05$.

RESULTS

Crab consumption of broadcast rodenticide bait at Palmyra Atoll

Both bait application rates used to treat the study sites at Palmyra failed to reach all rats from each sample population. Of the rats sampled on Whippoorwill and Portsmouth Islands (treated with 18 kg/ha + 9 kg/ha), 32% (29/91) and 3% (1/31) respectively lacked external or internal biomarker sign. On Home Island (treated with 36 kg/ha + 36 kg/ha), 5% (1/21) of captured rats lacked biomarker sign.

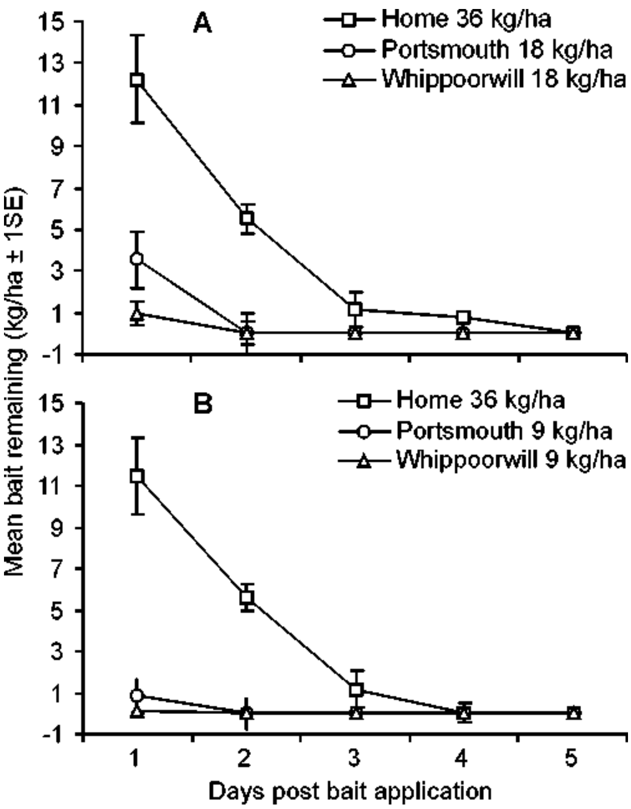

Fig. 2 Daily bait consumption estimates for the first "A" and second "B" bait applications. Mean consumption was measured by observing bait removal from fixed 25 x 1 m plots: Home = 22 plots, Portsmouth = 7 plots, Whippoorwill = 18 plots. Error bars represent ± 1 standard error of the sample mean.

Table 1 One-way ANOVA results for a between-site comparison of mean number of pellets removed from bait consumption plots by days post bait application for both the first and second bait application.

Bait application	Days post application	F-statistic	P-value
first	1	4.18	0.013
	2	11.84	< 0.001
	3	8.4	0.001
	4	2.8	0.072
	5	1.61	0.211
second	1	30.39	< 0.001
	2	26.7	< 0.001
	3	11.71	< 0.001
	4	3.77	0.031
	5	2.74	0.076
	6	1.81	0.176

Lower application rates (9 kg/ha and 18 kg/ha) resulted in rapid bait consumption. Following the 9 kg/ha application on Whippoorwill, only two of 180 pellets (4.6g of 414g of bait) remained in 18 bait consumption plots after a 24 hour period; the two remaining pellets were consumed by day two (Fig. 2) and bait was not observed elsewhere on the island. Similar bait consumption rates were observed on Portsmouth with nearly all of the 9 kg/ha application consumed within 24 hrs, and all pellets within bait consumption plots consumed within three days. The 18 kg/ha baiting regimes on Portsmouth and Whippoorwill resulted in more bait remaining after the initial 24 hours. However, there was complete consumption of baits in all plots on both islands within three days of the bait application. The 36 kg/ha bait applications at the Home study site were followed by 60% bait consumption within 24 hours, and small amounts of bait persisting through day five in the first application, and day six in the second application. Comparisons of daily mean bait consumption between all three study sites showed that Portsmouth and Whippoorwill had similar bait consumption patterns for 18 kg/ha and 9 kg/ha bait application regimes. However, mean bait consumption at the Home site was significantly higher than recorded at Portsmouth or Whippoorwill for days 1 through 3 after the first bait application, and days 1 through 4 after the second bait application (Table 1).

Land crab IOA rankings varied between study sites (Table 2). Home Island had relatively fewer land crabs than Portsmouth or Whippoorwill, and Whippoorwill had the highest crab density ranking.

Internal signs of biomarker from baits eleven days after the first bait application (five days after the second bait application) were 90% for hermit crabs from Home, 80% for Whippoorwill, and 70% for Portsmouth. When the mean number of crabs with and without internal or external biomarker sign was compared between the three study sites, there was no significant between-site

Table 2 Index of Abundance (IOA) ranking for land crab species at the three study sites; 1 = lowest density, 3 = highest density. If the species densities are similar between study sites, the study sites received the same ranking. Estimated mean density of crabs throughout Palmyra Atoll is 296 ± 139 (1SD) crabs/ha (Howald *et al.* 2004).

Study site	*Coenobita perlatus*	*Coenobita brevimanus*	*Cardisoma carnifex*	*Cardisoma rotundum*
Home	1	1	1	1
Portsmouth	2	1	1	2
Whippoorwill	3	2	1	1

Table 3 *Rattus exulans* day and night habitat use patterns on Dekehtik Island (9-13 February 2007), Pohnpei, FSM.

Rat ID	Active?	Day Ground (%)	Day Tree (%)	Night Ground (%)	Night Tree (%)
1	No	0	75	6	19
	Yes	33	25	33	8
2	No	50	40	0	10
	Yes	22	0	78	0
3	No	75	8	17	0
	Yes	0	33	67	0
4	No	46	31	8	15
	Yes	30	10	60	0
5	No	41	47	6	6
	Yes	55	0	36	9
6	No	67	33	0	0
	Yes	100	0	0	0
7	No	21	79	0	0
	Yes	31	8	46	15

difference in the number of crabs without external (F = 0.6, P = 0.554) or internal (F = 0.51, P = 0.606) biomarker sign, despite differences in the amount of bait applied to Home, Whippoorwill and Portsmouth Islands.

Habitat use by rats on Dekehtik Island

All study rats on Dekehtik were predominantly active at night, and while active, were more frequently observed on the ground rather than in the forest canopy. Daytime observations found most rats in the forest canopy and inactive (Table 3). Over 60% of the observations found the study rats in the crowns of coconut palms while in the forest canopy (Table 4), yet coconut palms only account for 30% of Dekehtik's forest canopy area (Wegmann *et al.* 2007).

DISCUSSION

Land crab interference with eradication outcomes

Failed eradications using bait stations against rodents on tropical islands have been attributed to interference of bait stations by crabs (Howald *et al*. 2004; Wegmann 2008). Baits that are broadcast side-step the limitations of bait station operations by minimising search time and handling time for target animals. However, bait broadcast faces a potentially insurmountable bait competition scenario when conducted on islands with large populations of crabs. Conventionally, bait application rates are set high enough to allow every rat four days of exposure to baits (Howald *et al*. 2007; Wegmann 2008). Yet on islands with land crabs, this convention would require excessively high bait application rates.

At Palmyra, even with two bait applications of 36 kg/ha, which is three times more bait than is typically applied to temperate islands (Veitch and Clout 2002; Towns and Broome 2003), not all rats were exposed to baits but they were accessible to nearly every crab. Furthermore, an increase in bait consumption was observed during the second application at Home Island, in the absence of rats (n = 15), which had been removed from the population during the first biomarker sampling session (Fig. 2). This suggests that at Palmyra, and presumably on islands with similar crab communities, crab related bait consumption prevents repeat bait applications ≥ 5 days apart from having a cumulative effect on the amount of bait made available to targeted rodents. To be effective, the bait application rate for the second broadcast should be the same as that of the first.

Also, for the two sites treated at 18 kg/ha + 9 kg/ha, bait consumption was greater on Whippoorwill Island, which had the highest crab IOA. Across the three study sites, bait consumption was greatest on Home Island, which had the highest bait application rate. These results suggest that crab abundance and bait application rates have a positive correlation with bait consumption. This is not a linear function because an increase in bait application rate leads to an increase in bait consumption – to an unspecified asymptote.

Bait-broadcast technology has allowed resource managers to effectively remove invasive rodents from islands that are too large or too topographically complex for bait-station-based operations (Howald *et al*. 2007). Also, bait application rates can likely be increased to meet the threshold for 100% exposure of all rats on crab-rich islands. However, doing so could increase the rodenticide exposure risk for non-target species, such as land birds and terrestrial mammals (Godfrey 1984; Eason and Spurr 1995; Hoare and Hare 2006). Future research on eradicating rodents from tropical islands should include the development of strategies or tools that will greatly reduce risk to non-target species while maintaining a high probability of eradication success. For example, crab deterrent compounds, that have neutral or positive palatability value for the targeted rodents, incorporated in the matrix of rodenticide bait could significantly lower requisite bait application rates for tropical island eradication campaigns.

Rat use of complex, 3D habitat

All seven radio collared rats were observed both on the ground and in the forest canopy during the four day study period. This habitat use pattern is similar to that of *R. rattus* at Palmyra Atoll (Howald *et al*. 2004) and enforces the understanding that rats on tropical islands function in a three-dimensional habitat that includes the forest canopy. When baits are broadcast by hand or from the air, it is essential to use applications rates that are high enough to ensure that every rat has access to a lethal dose of bait. However, given the documented use of the forest canopy by both *R. rattus* and *R. exulans*, it is equally important to include the forest canopy in the baiting plan; this happens naturally with an aerial broadcast, and can easily be incorporated into a hand-broadcast approach through sling-shot canopy baiting (Wegmann *et al*. 2007; Wegmann *et al*. 2008). All radio-collared rats on Dekehtik utilised both ground and canopy habitat, and most were more active while on the ground, which suggests that most foraging is on the ground. However, bait applied to the forest canopy, either by hand or through aerial application, is not available to crabs, and consumption by rats of bait placed in the forest canopy has been observed (Wegmann *et al*. 2007). Because of this, canopy baiting can be considered an "insurance" measure on crab-rich islands. Furthermore, the demonstrated preference for coconut palm canopy habitat by *R. exulans* on Dekehtik, also documented with *R. rattus* at Palmyra atoll (Howald *et al*. 2004), supports targeting coconut palms while canopy baiting.

Table 4. Tree species preferred by *R. exulans* utilising forest canopy habitat on Dekehtik Island, Pohnpei State, Federated States of Micronesia.

Tree species	% of total observations
Barringtonia asiatica	4
Cocos nucifera	66
Ficus sp.	20
Guettarda speciosa	2
Morinda sp.	4
Pandanus sp.	5

CONCLUSION

The risk of land crab interference and canopy preference to eradication success can be mitigated through increased broadcast application rates and aerial broadcast techniques. Such practices require close scrutiny for potential adverse impacts to non-target species. A better understanding of the eradication challenges inherent to tropical environments may enhance the conservation community's ability to effectively and safely manage the threat that invasive rodents present to tropical island biodiversity.

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Consideration of rat impacts on weeds prior to rat and cat eradication on Raoul Island, Kermadecs

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Abstract In anticipation of the planned eradication of rats (*Rattus norvegicus* and *R. exulans*) and cats (*Felis catus*) on Raoul Island in 2002, the exotic flora was evaluated to determine which species might become more invasive following the removal of rats. The interactions between the three mammal species targeted for eradication and the multiple weed species on the island were considered. A group of exotic species that had expanded their range vegetatively but not been observed fruiting in the presence of rats was identified. This group included grape (*Vitis vinifera*), shore hibiscus, fou (*Hibiscus tiliaceus*), rosy periwinkle (*Catharanthus roseus*) and airplant (*Bryophyllum pinnatum*). As a precaution, grape was targeted for eradication as this species would be dispersed effectively by tui (*Prosthemadera novaeseelandiae*), a native honeyeater, and blackbird (*Turdus merula*) if it began to fruit after rats were eradicated. Grape proved difficult to control but by mid-2002 all nine known grape infestations were reduced to zero density. In 2008/2009, no grape sprouts were found during searches of all known infestation sites. Since the eradication almost all species that did not fruit when rats were present are now fruiting.

Keywords: *Rattus exulans*; *Rattus norvegicus*; *Felis catus*; invasive weeds; *Vitis vinifera*

INTRODUCTION

Plants and animals have been introduced to many islands around the world (Abbott *et al*. 2000; Sax and Gaines 2008; Towns *et al*. 2006). Raoul Island is no exception, having been colonised through human agency since the first Polynesian voyagers arrived in the 10th century AD (Anderson 1980). *Rattus exulans* Peale (Pacific rat, kiore) were well established in the mid 1800s and were assumed to be native (MacGillivray 1854). However, they are most likely to have been introduced by Polynesian voyagers many centuries before (Harper and Veitch 2006). *Felis catus* L.(cat) accompanied the earliest human settlers and were definitely present in 1854 (MacGillivray 1854). *Rattus norvegicus* Berkenhout (Norway rat) colonised Raoul Island after the schooner "Columbia River" was wrecked there in September 1921 (Harper and Veitch 2006).

Exotic plants deliberately introduced were food and utility species such as *Cordyline fruticosa* (L.) Goepp. (ti pore, ki) and *Aleurites moluccana* (L.) Willd. (candlenut), introduced by Polynesians, and a range of fruiting trees, pasture grasses, vegetables and flowering garden plants introduced by European settlers (Sykes *et al*. 2000). Many plant species were accidentally introduced including *Bidens pilosa* L. (beggar's ticks) with Polynesians, *Conyza bonariensis* (L.) Cronquist (fleabane) with early European settlers (Sykes *et al*. 2000) and more recently, *Selaginella kraussiana* (Kunze) A.Braun (selaginella), first recorded in 1999 (West 2002) and *Soliva sessilis* Ruiz and Pav. (Onehunga weed) first recorded in 2008 (David Havell pers. comm.). Raoul Island has been intermittently occupied by people since C. 960 A.D. and permanently occupied since 1937 (Sykes *et al*. 2000).

Raoul Island, a Nature Reserve managed by the Department of Conservation, is the northernmost (29°15' S, 177° 55' W) and largest (2934 ha) of the Kermadec Group, situated in the South Pacific Ocean north of New Zealand (Fig. 1). It is an active volcano with rugged topography, subject to cyclones in the summer months and occasional heavy rain that triggers landslips. The Raoul volcano last erupted on 17 March 2006. Forest dominated by pohutukawa (*Metrosideros kermadecensis* W.R.B.Oliv.) is the main vegetation cover. Grasses, strand vines and ferns dominate coastal cliffs and the beach ridges at Denham Bay and Low Flat.

Approximately 20% of the vascular plant flora is endemic (Sykes *et al*. 2000). This comparatively low level of endemism is principally due to the young age of the Raoul volcano coupled with its remoteness. The high degree of disturbance associated with active volcanism may also be a contributing factor. Many of the endemic plant species closely resemble species endemic to the New Zealand mainland and offshore islands e.g., Kermadec pohutukawa, Kermadec ngaio (*Myoporum kermadecense* Sykes) and Kermadec fivefinger (*Pseudopanax kermadecensis* (W.R.B.Oliv.) Philipson). Little is known about elements of the flora that might be now extinct as a result of human occupation or eruption history: pollen

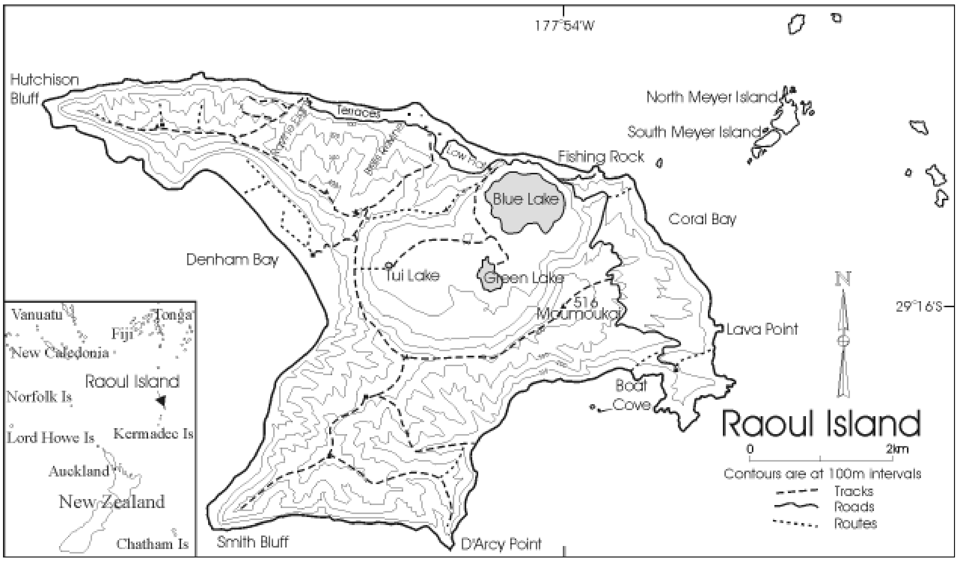

Fig. 1 Location of Raoul Island and location of places mentioned in the text.

Pages 244-247 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

analyses may shed some light on species turnover in this highly disturbed island ecosystem. The 2006 eruption has caused the extinction of at least one native species – *Ophioglossum petiolatum* Hook. (stalked adder's tongue) – that was known from a small area in Green Lake crater that is now both under water and buried in a thick mud deposit.

More than half the vascular plant flora of Raoul Island is introduced and a number of vines, trees and shrubs are transformer weeds (Pyšek *et al.* 2004). The majority of the introduced species are herbaceous and associated with disturbed ground along roads and open tracks, the accommodation for staff, meteorological station and abandoned rough pasture along the northern terraces of the island. There is an active surveillance programme, and this is how the selaginella and Onehunga weed were detected. Selaginella, in particular, could become a problem; however, prompt action has prevented this.

Weed eradication has been a focus of management on Raoul Island since 1972, with 29 species targeted for eradication in 1996 (West 2002). Goats were eradicated from Raoul Island in 1984 (Sykes and West 1996). The impact of goats on the native and exotic flora was evident both before and after they were eradicated: no exotic plant species are known to have increased significantly after goats were eradicated (Parkes 1984; Sykes and West 1996). Rats and cats were eradicated from Raoul Island in 2002 and 2004 respectively (Broome 2009), thus removing all introduced mammals. The need to identify the effect of rat eradication on the exotic flora was considered well before the rat eradication operation (West 2002; Sykes and West 1996) in order to avoid unintended outcomes (Zavaleta *et al.* 2001; Caut *et al.* 2009).

The effect of the Pacific rat on seeds and seedlings of native tree species of northern New Zealand islands has been documented (Campbell and Atkinson 1999; Campbell and Atkinson 2002; Towns *et al.* 2006). Rats were shown to eat seeds and seedlings, thus depressing the populations of at least 11 tree species. Norway rats also have been shown to suppress regeneration of native tree species by eating seeds and seedlings (Allen *et al.* 1994; Towns *et al.* 2006). Rats eat many plant parts including flowers, seeds, fruits and seedlings (Atkinson and Towns 2005; Innes 2005). There is no published information on impacts of rats on exotic plant species in New Zealand. However, the factors that predispose native species to predation by rats were assumed to apply to exotic species as well: seedlings of all species, and plants with fleshy fruit and/or fruit with large edible seeds were considered to be vulnerable (Towns *et al.* 2006).

Cats on Raoul Island primarily preyed upon rats, particularly the Pacific rat, and secondarily upon birds, some of which are effective seed dispersers (Fitzgerald *et al.* 1991). Rats also prey upon some of the same seed dispersing bird species, mainly introduced passerines (Towns *et al.* 2006). On Raoul, the principal fruit-eating and seed-dispersing bird species are tui (*Prosthemadera novaeseelandiae* Gmelin), blackbird (*Turdus merula* L.), thrush (*T. philomelos* Brehm) and starling (*Sturnus vulgaris* L.).

Ahead of the mammal eradication, therefore, direct and indirect effects of rats and cats on population dynamics of weeds were evaluated. Direct effects of rats that were considered were predation of flowers, seeds, fruit and seedlings. Indirect effects were predation by rats and cats on seed dispersing bird species.

METHODS

A food web describing the interactions observed between the predominant species and guilds on Raoul Island was constructed, based on published information and observations (Fig. 2). Particular attention was paid to known prey and competitors of rodents and cats and those organisms known to eat or disperse seeds. It was considered that the plant eradication regime on Raoul was resourced sufficiently to be able to cope with any increase in the abundance or range of the target species if they responded positively to rat eradication, since a key action in achieving eradication is to detect all individuals before they produce ripe fruit/seeds (West 2002).

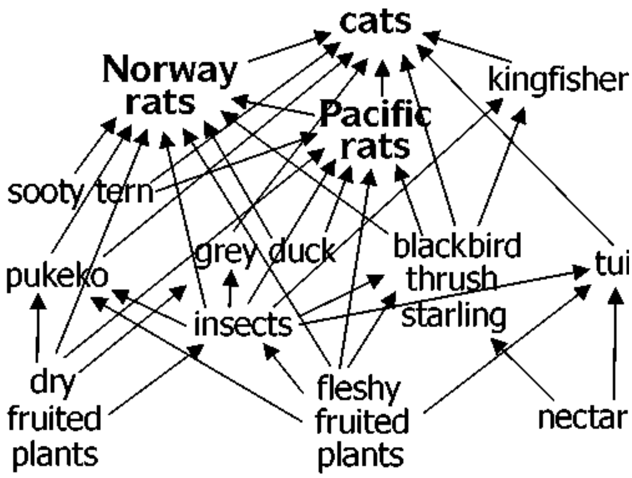

Fig. 2 Raoul Island food web from 1984–2002. Relationships derived from published sources (Fitzgerald *et al.* 1991; Harper and Veitch 2006) and from observations (C.R. Veitch pers. comm., and pers. obs.).

The list of exotic vascular plant species present on Raoul Island, excluding those currently targeted for eradication or control of seedlings (species of historic value), was reviewed to identify those that might become invasive after rats were eradicated. The eradication of goats showed that the primary vegetation response was a significant increase in density and abundance of native plant species, indicating that forest regeneration was not impaired by the majority of the exotic species which are herbaceous (a full list of exotic species can be seen in Sykes *et al.* 2000). Woody trees and shrubs, vines and clonal semi-woody perennials pose the greatest threat to the forest communities on Raoul Island. The primary question, therefore, was would a species, if not checked through seed or seedling predation, expand its population to the detriment of native plant communities on the island? Many species were eliminated at this point, particularly light-demanding, herbaceous species. Non-forest communities, where light-demanding weeds might thrive, are in comparatively harsh environments (e.g., coastal cliffs, rocky ridges, heated ground in the crater, back dunes) and have not indicated susceptibility to weed invasion to date, with the exception of the dune slack at Denham Bay where airplant has spread vegetatively.

The reproductive status and dispersal potential of the remaining species were then considered. Gravity or wind-dispersed species of short stature were considered to pose lesser risk as any progeny would be readily located near parent plants (Clark *et al.* 2005). Bird dispersed, woody species were identified as the group posing the greatest risk as their seeds could be dispersed over longer distances and their seed shadow would be poorly defined (Gosper *et al.* 2005). It would be difficult to find any progeny, and populations of these species would be likely to expand.

In examining the exotic flora, it was clear that five species were not regenerating from seed: all expansion was by vegetative means alone. In addition, fruit had been rarely (*Hibiscus tiliaceus* L.) or never observed on these species in the decade leading up to the rat eradication.

Therefore, the potential for each of these species to bear fruit and regenerate via seedlings was investigated.

RESULTS

Two introduced tree species that flowered and fruited freely were targeted for eradication prior to the rat and cat eradication. *Vitex lucens* Kirk (puriri), a fleshy-fruited tree, native to New Zealand, was represented on Raoul by three trees planted by European settlers. On Tiritiri Matangi Island in the Hauraki Gulf near Auckland, Pacific rats had suppressed all regeneration of the two puriri trees native to the island (pers. obs.). Puriri has fleshy fruits, the smallest of which can be dispersed by tui. The three trees on Raoul were felled in 1997 and did not regenerate. New, mature, trees have since been reported in a new location and are being investigated. A small group of *Macadamia tetraphylla* L. Johns. (macadamia) trees was felled in the same year and three seedlings were pulled out from the same site in 2003 (West 2002). Although the macadamia trees produced heavy nut crops in some years, the rats on Raoul Island efficiently devoured all seeds as evidenced by the many rat-gnawed shells beneath the trees.

The five species that were not known to fruit in the presence of rats were *Vitis vinifera* L. (grape), *Hibiscus tiliaceus* L. (shore hibiscus, fou), *Catharanthus roseus* (L.) G. Don (rosy periwinkle), *Bryophyllum pinnatum* (Lam.) Oken (airplant), and *Phoenix dactylifera* L. (date). Firstly, the reproduction of these species was investigated to determine what factors might limit them (Table 1). Dates are dioecious and the one clump of dates at Denham Bay is likely to be either male or female. Although the date flowers periodically, flowers have never been examined to determine the gender of the trees. Thus dioecy, with just one gender present, probably limits sexual reproduction of the date. For the remaining four species no reproductive limitations were detected other than the possibility of rat predation on developing fruit or, in the case of grapes, possibly the flowers.

The risk posed by the four species potentially limited by rat predation was evaluated (Table 2). Grapes were found to be the only high risk species: there were nine separate plants growing in a range of locations and they are bird dispersed. Therefore they could arise at locations some distance from parent plants. One of the grape vines occupied an area of C. 0.5 ha at Low Flat and another in Denham Bay was estimated to be 4 ha in extent (C.R. Veitch pers. comm.). Grapes were formally targeted for eradication in 1998 as a result of this analysis. Initially Grazon (triclopyr-based herbicide) and Roundup (glyphosate) were used to kill grape vines. Later Vigilant (a gel formulation of picloram) was used. Grapes took considerable effort to kill but by 2008–09 just two grape sprouts were found (Fig. 3) and none were ever observed flowering or fruiting (they were killed before getting to this stage).

The other three species not targeted for eradication – shore hibiscus, rosy periwinkle and airplant all began to fruit after rats were eradicated. Seedlings are commonly observed near parent plants. Airplant control began in 2007

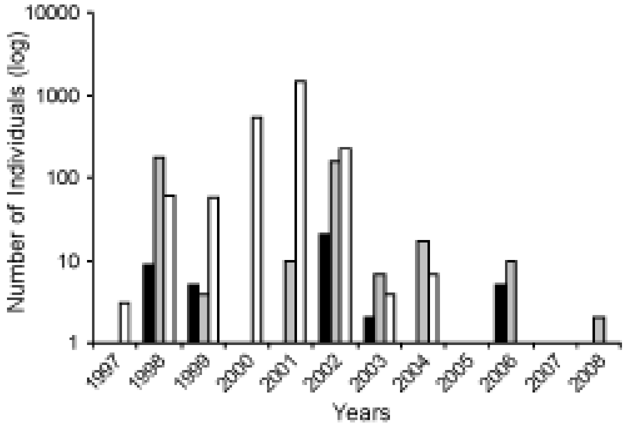

Fig. 3 Number of individuals of grape in different size classes removed from Raoul Island from 1997 to 2008: small (< 30 cm tall: black bars); medium (30-100 cm tall: grey bars) and large (>100 cm tall: white bars).

and rosy periwinkle control in 2008. Control of both species is aimed at containing the populations so they don't spread beyond their former extent. Both species are confined to Denham Bay with airplant intermittently spread along 1–2 km of the dune slack. Rosy periwinkle occupies a smaller area at the forest edge. Shore hibiscus is currently not controlled. Further investigation of its status is required as the plants in Denham Bay and at Low Flat are thought to be introduced whereas a plant at Coral Bay is likely to be native (West 1996).

Two unexpected outcomes have been observed. Firstly, when a large grape vine in the canopy of pohutukawa and other trees was eradicated from an old (19th century) garden site at the south end of Denham Bay, a dense stand of the introduced canna lily (*Canna indica* L.) arose. Canna lily was first recorded on Raoul Island only after eradication of another weed (Mysore thorn – *Caesalpinia decapetala* (Roth) Alston) began (Sykes and West 1996). Establishment of the canna lily from seed that had lain dormant in the soil for perhaps a century was facilitated by increased light levels after grape eradication and the absence of rat predation.

Secondly, aroid lily (*Alocasia brisbanensis* (F.M. Bailey) Domin), a widespread weed on Raoul which was suppressed by canopy closure after the goat eradication, is now patchily heavily browsed by the tropical army worm (*Spodoptera litura* Fabricius). The type of browsing damage now seen on the aroid lily was never observed prior to the rat eradication. The aroid lily appears to be the only plant obviously being browsed by this noctuid caterpillar.

DISCUSSION

Some studies have indicated that exotic plants can become invasive after herbivore or seed predator removal (Caut *et al*. 2009; Zavaleta *et al*. 2001; Abbott *et al*. 2000). These effects were anticipated on Raoul Island and the

Table 1 Reasons for non-fruiting of exotic plant species on Raoul Island.

Species	Pollinator	Climate	Dioecious	Rat browse
Grape	wind	suitable	no	?
Shore hibiscus	insects	suitable	no	?
Rosy periwinkle	insects	suitable	no	?
Airplant	insects	suitable	no	?
Date	wind	suitable	yes	?

Table 2 Risk posed by each of the species that could have fruited but did not when rats were present on Raoul Island.

Species	Distribution	Fruit type	Disperser	Invasion risk
Grape	9 sites	fleshy	tui, blackbird	high
Shore hibiscus	2 sites, inland	capsule	gravity, sea	low
Rosy periwinkle	1 site	capsule	gravity	low
Airplant (no plantlets)	1 site	capsule	wind	low

exotic flora was scrutinised for species that could change their status following rat and cat eradication. Just three species were identified that warranted eradication prior to the mammal eradication operation. Despite grapes not being recorded as an invasive species outside of North America (Reichard 1994), this species possessed sufficient characteristics (Buckley *et al*. 2006) to indicate it could become a widespread transformer once it was released from suppression by rat predation and available for dispersal by native and exotic birds which were also released from predation by rats and cats.

Although the evidence of direct effects of rats on fruiting and seedling production of the five weed species investigated is circumstantial, the response of airplant, rosy periwinkle and shore hibiscus to release from rat predation is sufficiently striking to suggest that grapes also were impacted by rats and their eradication was justified.

The eradication of Pacific rats, Norway rats and cats from Raoul Island has changed the ecosystem from one dominated by predators to one where competition for resources is the primary driver (Broome 2009). Seabirds are returning, and so have red-crowned parakeets (*Cyanoramphus novaezelandiae cyanurus*) (Ortiz-Catedral *et al*. 2009) and spotless crakes (*Porzana tabuensis*) (pers. obs). Invertebrates are prominent whereas before they were not.

Rats are seed dispersers as well as seed destroyers (Bourgeois *et al*. 2005) and cats can also disperse seeds (Bourgeois *et al*. 2005; Nogales *et al*. 1996). However, they are functionally replaced, to a degree, on Raoul Island by the return of red-crowned parakeets and an increase in other resident bird and insect species. Birds and insects will now be the species that disperse and destroy seeds on plants, on the ground and in the seed bank, as well as seedlings. Reintroduction of fruit pigeons (*Hemiphaga novaeseelandiae*) which previously occurred on Raoul, will add to the suite of dispersers of larger fruited native plants, such as the palm *Rhopalostylis baueri.*

Unanticipated or surprise effects arising from the rat and cat eradication detected so far are relatively minor and in the case of tropical army worm defoliating aroid lily somewhat beneficial. Monitoring of changes post-eradication is essential (Zavaleta *et al*. 2001) in order to be able to respond to any detrimental changes (Mack and Lonsdale 2002).

Over time, and with the continuation of the weed eradication programme, the ecosystem may be restored to an approximation of its state before Polynesians arrived. The remoteness of Raoul Island (1100 km NNE of New Zealand) may be sufficient to reduce the likelihood of weed invasion increasing as seabirds return (Mulder *et al*. 2009), so long as weeds continue to be managed, surveillance is programmed, and island quarantine procedures are enforced.

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Results and Outcomes

The results of work demonstrating that the invasive species has been successfully eradicated; and outcomes of eradications, particularly recording responses by native species.

Island restoration in Mexico: ecological outcomes after systematic eradications of invasive mammals

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Abstract On Mexican islands, 20 island endemic species and subspecies of vertebrates have gone extinct in the last 100 years; all but four of these extinctions were caused by invasive mammals. To prevent more extinctions, 49 populations of 12 invasive mammals were eradicated from 30 Mexican islands. These actions protected 202 endemic taxa – 22 mammals, 31 reptiles, 32 birds, and 117 plants – as well as 227 seabird breeding colonies. An area of 50,744 ha was restored, which represents 10% of Mexico's total island territories. Techniques have ranged from the traditional – trapping and ground hunting – for 26% of the restored area, to the most sophisticated – aerial hunting, aerial broadcast of bait, DGPS and GIS use – for 74% of the restored area. These conservation actions are of high significance for Mexico. Extirpated seabirds have recolonised several islands and increased reproductive success has been documented. An ongoing seabird social attraction project facilitates recolonisation of additional islands. On Guadalupe Island, after the eradication of goats (*Capra hircus*), recruitment of three endemic trees increased from zero to more than 150,000 individuals. Six native plants, including two endemics, were rediscovered. Ecological outcomes from island restoration are expected to increase. Eradicating all invasive vertebrates from the remaining 41 Mexican islands with 83 populations of invasive mammals is a viable and strategic goal, achievable by 2025, and will set a global benchmark.

Keywords: Eradication, restoration, invasive mammals, islands, Mexico, ecological outcomes, strategic goal.

INTRODUCTION

Invasive alien species are considered second behind habitat destruction as the largest cause of biodiversity loss worldwide (Courchamp *et al.* 2003). When islands are considered alone, invasive species are the primary cause of extinctions (Baillie *et al.* 2004; Reaser *et al.* 2007). According to Ebenhard (1988), invasive species can affect native species through: 1) modification of plant populations and the animals that depend on them; 2) predation; 3) competition for local resources; 4) dispersal of micro and macro parasites; 5) genetic changes in native populations through hybridisation; and 6) prey of native predators (changing the food chain). Over time, these impacts can restrain recruitment, cause the extinction of species and modify food webs and ecological processes. Insular populations are particularly vulnerable since they have not co-evolved with invasive species and lack defence mechanisms against them (Primack 2002).

Mexican islands comprise a total area of 5127 km^2 (INEGI 2005). The northwest region is particularly important, where more than 600 islands in the Pacific Ocean off Baja California, the Gulf of California, and the Revillagigedo Archipelago significantly contribute to Mexico's megadiversity. These islands have more endemic plants and vertebrates than the Channel Islands of the USA and the Galapagos Islands of Ecuador. Compared with the endemic taxa of the Galapagos Islands (310), the northwest Mexican islands (331) have 25% more endemic species per km^2. The Mexican islands are crucial nesting and resting sites for seabirds and pinnipeds, as well as important refuges for harvested marine species that have been over-exploited on the coastal mainland. Considerable conservation and restoration efforts, especially to overcome the effects of invasive species, have been invested in Mexican islands. Most of this has been in the past two decades, with close and practical collaboration between federal government, NGOs and academic institutions.

Under the Mexican constitution all islands are part of national territory and are under federal jurisdiction, except for a few islands that are in the jurisdiction of individual states (Moreno-Collado 1991; Cabada-Huerta 2007), and very few which are communal or private property (CONANP-SEMARNAP 2000). Federal islands are administered by the Ministry of the Interior (SEGOB) and protected by the Ministry of the Navy (SEMAR). For more than 15 years, control and eradication projects have been undertaken in these territories through interpretations of the Environmental Act (LGEEPA) and the Wildlife Act (LGVS), under the mandate of the Ministry of the Environment (SEMARNAT).

In this paper we review the history of invasive vertebrate introductions to Mexican islands, the impacts of the introduced species, current progress with reversing these effects, and the responses of native species when introduced species are removed. We use Guadalupe Island to illustrate the processes of extinction and the prospects for recovery. We also update the available information on invasive vertebrates on Mexican islands.

In Mexico, island biodiversity has been seriously affected by introduced invasive species but these effects were not studied until the 1980s (Mellink 1992, 1993; Velarde and Anderson 1994). Subsequently, the situation on northwest Mexican islands as a result of introduced rodents was described by Romeu (1995) as critical. The first eradication projects were implemented in 1994 and 1995, on Asunción and San Roque islands off the Baja California Peninsula (Aguirre-Muñoz *et al.* 2008), and on Rasa Island, in the Gulf of California (Ramírez-Ruiz and Ceballos-González 1996).

More recently, research on the status and impacts of invasive vertebrates has been published and lists of the distribution of invasive species generated (Mellink 2002; Aguirre-Muñoz *et al.* 2005; Aguirre-Muñoz *et al.* 2009a; Rodríguez-Malagón 2009). As part of this review we have checked, and in some cases corrected, existing datasets, with the result that tables presented here vary from those previously published.

HISTORY OF INVASIONS OF MEXICAN ISLANDS

Flora and fauna have been moved between locations for as long as people have moved around the world. On the islands of northwest Mexico before the 20th century, introductions of invasive species were largely related to the harvesting of marine mammals and mining for guano. Subsequently, the sources of introductions diversified to include commercial and sport fishing. Examples include house mice (*Mus musculus*), presumably introduced to Guadalupe Island during marine mammal hunting trips

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Table 1 Invasive mammals still present on Mexican islands in 2010.

Island	Area (ha)[†]	Common names
GULF OF CALIFORNIA		
Alcatraz (Pelícano)	50	House mouse
Almagre Chico	10	Ship rat
Ángel de la Guarda	93,068	Cat, house mouse, ship rat
Carmen	14,461	Goat, cat, bighorn sheep
Cerralvo	13,505	Goat, cat
Coyote	25	Dog, cat
El Rancho	232	House mouse, ship rat
Espíritu Santo	7991	Goat, cat
Granito	27	Ship rat
María Madre	14,388	Goat, cat, ship rat, horse, rabbit
María Magdalena	6977	Goat, white-tailed deer, cat, ship rat
María Cleofas	1963	Goat, cat, ship rat
Mejía	245	House mouse, ship rat
Melliza Este	1	Ship rat
Pájaros	82	Ship rat
Saliaca	2000	House mouse, ship rat
San Diego	56	Goat
San Esteban	3966	Ship rat
San José	18,109	Goat, donkey, cat
San Marcos	2855	Goat, cat
San Vicente	14	House mouse
Santa Catalina (Catalana)	3890	Northern Baja California deer-mouse
Tiburón	119,875	Dog, bighorn sheep
Total 1	303,790	
GULF OF MEXICO AND CARIBBEAN SEA		
Cayo Norte Menor	15[‡]	Ship rat
Cayo Norte Mayor	29[‡]	Ship rat
Cayo Centro	537[‡]	Ship rat, cat
Cozumel	47,000	House mouse, ship rat
Holbox	5540	Ship rat
Muertos	16[‡]	House mouse
Mujeres	396	House mouse, ship rat
Pérez	11[‡]	Ship rat
Pájaros	2[‡]	House mouse
Total 2	53,546	
PACIFIC		
Cedros	34,933[‡]	Dog, goat, cat, house mouse, ship rat, donkey
Clarión	1958	Rabbit
Coronado Sur	126[‡]	House mouse
Guadalupe	24,171[‡]	Cat, house mouse
Magdalena	27,773[‡]	Dog, donkey, cat, house mouse
Natividad	736[‡]	White-tailed antelope squirrel
San Benito Oeste	364[‡]	Cedros Island cactus mouse
Santa Margarita	21,504[‡]	White-tailed antelope squirrel, dog, goat, donkey, horse, cat
Socorro	13,033[‡]	House mouse, cat
Total 3	124,598	
Total 1+2+3	481,934	

Names of species listed in Table 1

Common name	Scientific name
Dog	*Canis lupus familiaris*
Bighorn sheep	*Ovis canadiensis mexicana*
Cat	*Felis catus*
Cedros Island cactus mouse	*Peromyscus eremicus cedrosensis*
Dog	*Canis lupus familiaris*
Donkey	*Equus asinus*
Goat	*Capra hircus*
Horse	*Equus caballus*
House mouse	*Mus musculus*
Northern Baja California deer-mouse	*Peromyscus fraterculus*
Rabbit	*Oryctolagus cuniculus*
Ship rat	*Rattus rattus*
White-tailed antelope squirrel	*Ammospermophilus leucurus*
White-tailed deer	*Odocoileus virginianus*

[†] INEGI (2005), unless indicated otherwise.
[‡] Area estimated by Conservación de Islas through satellite imagery (Samaniego-Herrera *et al.* 2007; Conservación de Islas-CONANP 2009).

(Moran 1996); and ship rats (*Rattus rattus*) and house mice, which probably arrived on Cedros Island with skin hunters (Mellink 1993). In association with guano mining, invasive rodents were introduced to San Roque, San Jorge, Rasa, and San Pedro Mártir islands, among others (Knowlton *et al.* 2007). The common house gecko (*Hemidactylus frenatus*) was introduced to Socorro and María Madre islands, probably with food supplies from the mainland (Valdez-Villavicencio and Peralta-García 2008), whilst common blind snake (*Ramphotyphlops braminus*) was probably introduced to Isabel Island with invasive fruit plants that were once present there (A. Samaniego-Herrera, pers. obs.). Introductions continue, with the spread of ship rats to Mejía Island and house mice to Coronado Sur Island, in the last 5 to 10 years. The Cedros Island cactus mouse (*Peromyscus eremicus cedrosensis*) was accidentally introduced in 2007 to the nearby San Benito Oeste Island.

A wide range of mammals have been intentionally introduced to Mexican islands. Dogs were taken to Guadalupe and Cedros islands as pets (Ibarra-Contreras 1995; Knowlton *et al.* 2007). As supplies of fresh meat, goats (*Capra hircus*) were introduced to Guadalupe, San Benito Oeste, Cedros, San José, Espíritu Santo and Cerralvo islands (Mellink 1993, 2002; Donlan *et al.* 2000; CONANP-SEMARNAP 2000), merino sheep (*Ovis aries*) to the tropical Socorro Island in the middle of the 18th Century; and sheep, pigs (*Sus scrofa*) and rabbits (*Oryctolagus cuniculus*) to Clarión Island (Everett 1988; Steve *et al.* 1991; CONANP-SEMARNAT 2004). To extend food reservoirs to the Seri tribe, chuckwallas (*Sauromalus* spp.) were introduced to Alcatraz Island (Case *et al.* 2002). For sustainable use as a sport hunting resource, bighorn sheep (*Ovis canadensis*) were introduced to Tiburón and Carmen islands (Álvarez-Romero and Medellín-Legorreta 2005), although they did not become invasive.

With the exception of bighorn sheep, none of the intentional introductions have brought the expected benefits. Furthermore, when domesticated species became feral and hard to hunt, they were replaced by food imported from the continent and feral populations grew without control. Table 1 shows in detail the invasive mammals still present on Mexican islands.

LOCAL IMPACTS OF INVASIVE VERTEBRATES

Introduced mammals on the islands of northwest Mexico have had major negative impacts on biodiversity, leading to extinction from these islands of 16 endemic species, including one – the Socorro Island dove (*Zenaida graysoni*) extirpated in the wild (Table 2), and now kept only in zoos. Four species listed by Aguirre-Muñoz *et al.* (2009a) are not in Table 2 because they probably did not become extinct due to introduced species. These include Mcgregor's house finch (*Carpodacus mexicanus mcgregori*) from San Benito Island, which presumably became extinct because of excessive collecting by scientists (Jehl 1970); Pemberton's deer mouse (*Peromyscus pembertoni*) from San Pedro Nolasco Island, which became extinct presumably because of competition with other native rodents (Flannery and Schauten 2001); the Guadalupe caracara or "quelele" (*Caracara lutosa*), last recorded in 1900 (Abbott 1933) probably due to excessive hunting and collecting of specimens combined with the indirect impacts of goats and cats (Jehl and Everett 1985; Stattersfield 1998); and the Turner Island woodrat (*Neotoma varia*) which was last seen in 1977, although there are no records of introduced species on Turner Island (Álvarez-Castañeda and Ortega-Rubio 2003).

Documentation and evaluation of impacts on Mexican islands has been limited, episodic and, in most cases, recent. Below, we summarise documented impacts of the most harmful and widely spread invasive species on Mexican islands.

Rodents

On Farallón de San Ignacio and San Pedro Mártir islands in the Gulf of California, isotopic analysis of ship rat diet allowed identification of those species most heavily affected by predation. On Farallón de San Ignacio, 90.4% of analysed rats fed exclusively on seabirds; whereas on San Pedro Mártir, consumption of plants, seabirds, and terrestrial and marine invertebrates was approximately equal (Rodríguez-Malagón 2009). This difference between islands reflected local food availability and confirmed the opportunistic and adaptable habits of this species of rat (Towns *et al.* 2006).

Cats

On Mexican islands, eight rodent taxa are extinct, or nearly so, and seven of these were probably due to predation by cats (*Felis catus*) (Table 2). Seabirds have been similarly affected, with the extinction of numerous island populations and total extinction of the Guadalupe storm-petrel (*Oceanodroma macrodactyla*) (Jehl and Everett 1985). The impact of cats was illustrated on Natividad Island where, before their eradication, 25 cats killed more than 1,000 black-vented shearwater (*Puffinus opisthomelas*) every month (Keitt *et al.* 2002).

Herbivores

Goats and sheep exert strong negative pressure on plant communities. They modify their species composition, which is often followed by soil erosion. They also compete with native herbivores (Parkes *et al.* 1996; Álvarez-Romero *et al.* 2008). In Mexico, goats had dramatic effects on the vegetation of Guadalupe Island (Moran 1996; Rodríguez-Malagón 2006; Luna-Mendoza *et al.* 2007), and also Espíritu Santo (León de la Luz and Domínguez-Cadena 2006), Cerralvo (Mellink 2002), and the Marías Archipelago (CONANP-SSP 2008). Sheep introduced to Socorro Island (Castellanos-Vera and Ortega-Rubio 1994) removed vegetation cover over most of the island and reduced habitat available for native birds (Rodríguez-Estrella *et al.* 1994).

RESPONSES TO INVASIONS

Island pest eradications

The eradication of invasive fauna on Mexican islands began in 1994-1995 with successful campaigns against feral cats on Asunción Island, feral cats and rats on San Roque Island (Aguirre-Muñoz *et al.* 2008), and rats and mice in Rasa Island (Ramírez-Ruiz and Ceballos-González 1996). Mammals remain the only vertebrate group eradicated from Mexican islands, with most of the successful examples using hunting, trapping, poisoning or a combination of these. Recently, radio-telemetry and trained dogs have been used. For large mammals, terrestrial and aerial hunting has been the most efficient

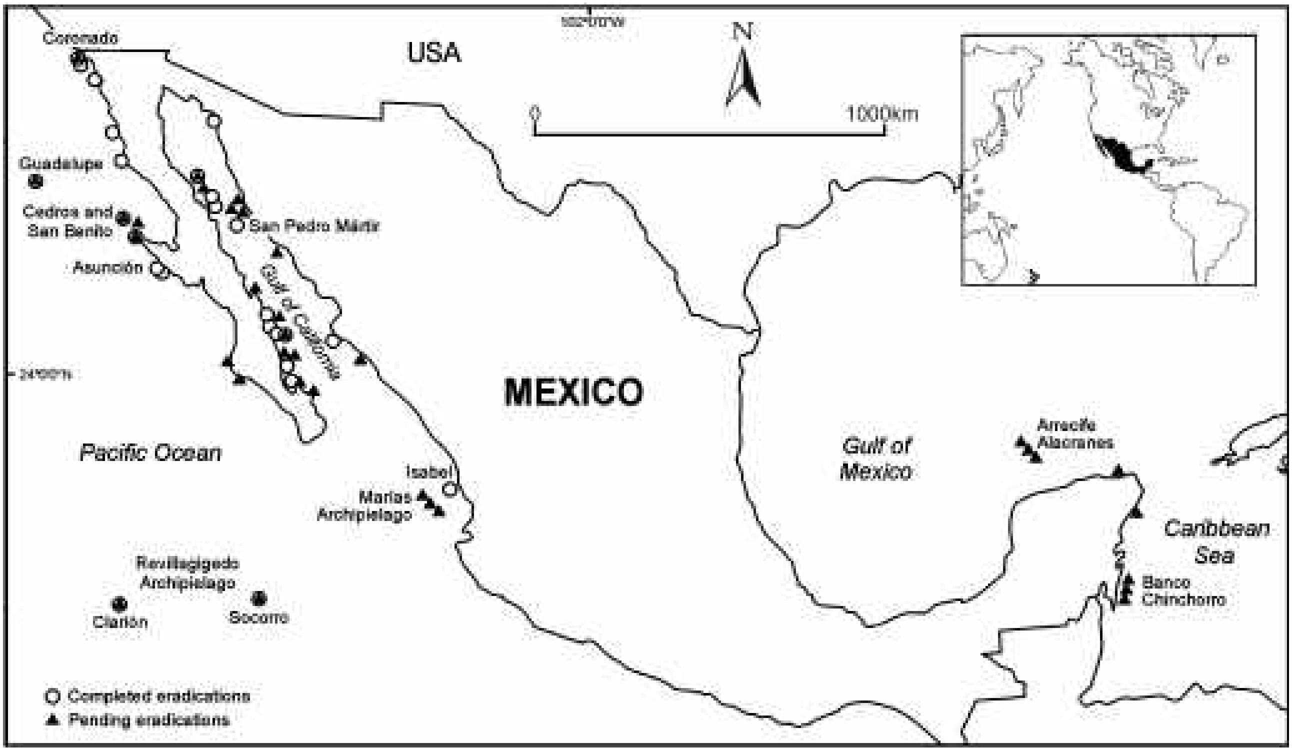

Fig. 1 Completed and pending eradications in Mexico between 1994 and 2010 (See Tables 1 and 3 for details).

Table 2 Likely extinctions of vertebrates after invasive species established on Mexican islands.

Species	Common name	Island	Year of last record	Year of last field search	Invasive species implicated and status	IUCN Category[†]
Birds						
Oceanodroma macrodactyla	Guadalupe storm-petrel	Guadalupe	1912 [A]	2000 [A]	Cat (SP), goat (ER) [1]	CR
Caracara lutosa	Guadalupe caracara	Guadalupe	1900 [B]	2003 [A]	Cat (SP), goat (ER) [1]	EX
*Zenaida graysoni**	Socorro dove	Socorro	1972 [O]	1981 [C]	Cat (SP), Sheep (SP)[1]	EW
Micrathene whitneyi graysoni	Elf owl	Socorro	1932 [D]	1981 [D]	Cat (SP) Sheep (SP) [1]	NE[‡]
Colaptes auratus rufipileus	Northern flicker	Guadalupe	1906 [B]	2003 [A]	Cat (SP) Goat (ER) [1]	NE[‡]
Thryomanes bewickii brevicauda	Bewick's wren	Guadalupe	1892 [B]	2003 [A]	Cat (SP) Goat (ER) [1]	NE[‡]
Regulus calendula obscurus	Ruby-crowned kinglet	Guadalupe	1953 [B]	2003 [A]	Cat (SP) Goat (ER) [1]	NE[¥]
Pipilo maculatus consobrinus	Spotted towhee	Guadalupe	1897 [B]	2003 [A]	Cat (SP) Goat (ER) [1]	NE[‡]
Aimophila ruficeps sanctorum	Rufous-crowned sparrow	Todos Santos	1927 [N]	2005 [L]	Cat (ER) [2]	NE[‡]
Mammals						
Chaetodipus baileyi fornicatus	Bailey's pocket mouse	Montserrat	1975 [E]	2003 [K]	Cat (ER) [2]	NE[£]
Neotoma anthonyi	Anthony's woodrat	Todos Santos	1950s [H]	2005 [L]	Cat (ER) [2]	EX
Neotoma bunkeri	Bunker's woodrat	Islas Coronado	1980s [E]	1997 [E]	Cat (ER) [2]	EX
Neotoma martinensis	San Martin Island woodrat	San Martín	1925 [I]	2006 [L]	Cat (ER) [2]	EX
Oryzomys nelsoni	Nelson's rice rat	María Madre	1898 [J]	2002 [J]	Cat (SP) ship rat (SP)[4]	EX
Peromyscus guardia harbisoni	Angel de la Guarda deer mouse	Granito	1973 [G]	1999 [G,P]	Ship rat (SP) [5]	CR
Peromyscus guardia mejiae	Angel de la Guarda deer Mouse	Mejía	1973 [G]	1999 [G,P]	Cat (ER) [2]	CR
Peromyscus maniculatus cineritius	Deer mouse	San Roque	1960's [F]	2009 [M]	Cat (ER) Ship rat (ER) [3]	NE[±]

[†]IUCN 2010. IUCN Red List of Threatened Species. Version 2010.1. <www.iucnredlist.org>. Downloaded on 06 April 2010.
CR=Critically endangered; E=Extinct; EW=Extinct in the wild; NE=Not evaluated.
SP= Still Present; ER= Eradicated.
* Extinct in the wild but bred in captivity in Frankfurt, Germany.
[‡] Listed as extinct in the Official Mexican Norm NOM-059-SEMARNAT-2001 (DOF 06-03-2002).
[¥] Listed as probably extinct in the Official Mexican Norm NOM-059-SEMARNAT-2001 (DOF 06-03-2002).
[£] Listed as subject to special protection in the Official Mexican Norm NOM-059-SEMARNAT-2001 (DOF 06-03-2002).
[±] Listed as endangered in the Official Mexican Norm NOM-059-SEMARNAT-2001 (DOF 06-03-2002).
A= Barton *et al.* 2005; B= Jehl and Everett 1985; C=Jehl and Parkes 1983; D=Jehl and Parkes 1982; E=Álvarez-Castañeda and Ortega-Rubio 2003; F= Alvarez-Castañeda and Patton 1999; G=Mellink *et al.* 2002; H=Mellink 1992; I=Cortés-Calva *et al.* 2001; J=Ceballos and Oliva 2005; K=GECI 2003; L=Samaniego-Herrera *et al.* 2007; M=Félix-Lizárraga *et al.* 2009; N=Van Rossem 1947; O=CONANP-SEMARNAT 2004; P=Álvarez-Castañeda and Ortega Rubio 2003; 1= Aguirre-Muñoz *et al.* 2009a; 2=Ortega-Rubio and Castellanos-Vera 1994; 3=Nogales *et al.* 2004; 4=Donlan *et al.* 2000; 5= CONANP-SSP 2008.

technique, in combination with radio-telemetry. For small mammals such as cats and rabbits, the combination of hunting and trapping, supported by detection dogs, has been particularly effective. For rodents, aerial spread of rodenticide has proved to be the most effective practice (Samaniego-Herrera *et al.* 2009, 2011; Table 3).

Between 1995 and 2010, 49 invasive mammal populations have been eradicated from 15 islands in the Pacific Ocean and 15 in the Gulf of California (Table 3; Fig. 1). These restoration actions have protected at least 117 species of endemic plants, 85 species of endemic vertebrates, and more than 227 populations of seabirds over a total area of 50,744 ha (Fig. 2). Feral cats have been eradicated from 18 islands, rodents and rabbits from 14 islands, and ungulates from 8 islands. The most significant contribution has been the eradication of goats and sheep from Guadalupe and Socorro islands respectively (Fig. 2 and Fig. 3). Rodent eradications also contributed to

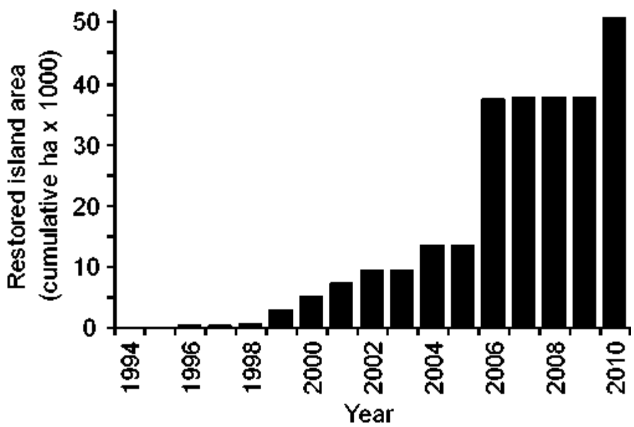

Fig. 2 Cumulative restored island area in Mexico from 1994 to 2010.

Table 3 Species, techniques and dates of eradication of invasive mammals from Mexican islands between 1994 and 2010.

Island	Area (ha)[£]	Species removed[‡]	Date of eradication	Methods	Year of last field search[‡]	References
Pacific Ocean						
Asunción	41	Cat	1994	Trap	2009	A,B,C,E,F,AA
Clarión	1958	Sheep, pig	2002	Hunt	2003	E,F
Coronado Norte	37[±]	Cats	1995-1996	Trap	2009	A,B,C,D,E,F
Coronado Sur	126[±]	Cat[1], goat[2] donkey	2003	Trap, hunt	2009	D,E,F,G,Z
Guadalupe[3]	24,171[±]	Rabbit, donkey	2002	Live removal*	2010	F,G
		Horse	2004	Live removal*		F,H
		Goat	2003-2006	Live removal*, trap, hunt and telemetry		F,I,J
		Dog[4]	2007	Live removal*, trap, hunt		E,F,I
		Goat, sheep	1997	Live removal*		A,D,E,F,K,U,V
Natividad	736[±]	Cat	1998-2000	Trap, hunt, live removal	2006	A,B,C,AB,AC
		Dog	2001	Live removal*		F
San Benito Este	146[±]	Rabbit	1999	Trap and hunt	2009	A,E,F,K,L,V
San Benito Medio	45[±]	Rabbit	1998	Trap and hunt	2009	A,E,F,K,L,V
San Benito Oeste	364[±]	Rabbit, goat	1998	Trap and hunt	2009	A,E,F,K,L,U,V
		Donkey*	2005	Live removal*		F,G
San Jerónimo	48[±]	Cat	1999	Trap and hunt	2006	B,C,D,F,K,AA
San Martín	265[±]	Cat	1999	Trap and hunt	2006	B,C,D,F,K,AA
San Roque	35	Cat[5]	1994	Trap	2009	A,B,C,D,E,F,K,AA
		Ship rat	1995	Bait stations		A,D,E,F,H,K,O,AD
Socorro	13,033[±]	Sheep	2010	Hunt and telemetry	2010	X
Todos Santos Norte	34[±]	Cat, rabbit	1999-2000	Trap and hunt	2009	A,B,C,E,F,K,V,AA,AC
		Donkey	2004	Live removal*		F,G
Todos Santos Sur	89[±]	Cat[¥]	1997-1998/ 1999/2004	Trap and hunt	2009	A,B,C,D,E,F,K,V
		Rabbit	1997	Trap and hunt	2009	A,B,C,D,E,F,K,V
Gulf of California						
Coronados	715	Cat	1998-1999	Trap	2008	B,C,K,M,N
Danzante	412	Cat	2000	Trap[€]	2008	C,F
Estanque	82	Cat	1999	Trap and hunt	2003	B,C,K,AA
Farallón de San Ignacio	17[±]	Ship rat	2007	Aerial broadcast	2009	F,P,AD
Isabel	80[±]	Cat[6]	1995-1998	Trap, hunt & bait stns	2009	A,B,C,E,F,K,Q
		Ship rat[7]	2009	Aerial broadcast		R, AD
Mejía	245	Cat	1999-2001	Trap and hunt	2005	B,C,E,F,K, AA,AB
Montserrat	1886	Cat[8]	2000-01/03	Trap and hunt	2008	B,C,E,F,K
Partida Sur	1533	Cat	2000	Live removal*	2007	B,C,E,F,K,AA,AB
Rasa[9]	57	Ship rat, house mouse	1995-1996	Bait stations	2009§	E,H,O,S,AD
San Jorge Este	9	Ship rat	2000-2002	Bait stations	2004	E,F,H,K,O,T,AD
San Jorge Medio	41	Ship rat	2000-2002	Bait stations	2004	E,F,H,K,O,T,AD
San Jorge Oeste	7	Ship rat	2000-2002	Bait stations	2004	E,F,K,T,AD
San Francisquito	374	Cat	2000	Trap and hunt	2005	B,C,E,F,K,AA
		Goat	1999	Hunt		F,U
San Pedro Mártir	267[±]	Ship rat	2007	Aerial broadcast	2009	F,P,AD
Santa Catalina (Catalana)	3890	Cat	2000-2004	Trap and hunt	2008	B,E,F,Y,Z
Total area	50,742					

[£] INEGI (2005), unless indicated otherwise; [±] Area estimated by Conservación de Islas through satellite imagery (Samaniego-Herrera *et al.* 2007; Conservación de Islas-CONANP 2009); [‡] Work conducted by Conservación de Islas unless indicated otherwise; *Small populations were removed alive; [§] E. Velarde. pers. comm. [€]During 2000 traps and track plots were set by CIBNOR's (Centro de Investigaciones Biológicas del Noroeste, S.C.) researchers to trap the feral cats. No cats were captured. However, during 2000 one cat was found dead on the island. Since then, no more tracks or signs have been recorded (Gustavo Arnaud pers. comm. 2010). [¥]Cats were reintroduced and eradicated in 1999 (Sánchez-Pacheco and Tershy 2000) and 2004 (Aguirre-Muñoz *et al.* 2004).

[1]First cat eradication: 2001 (Knowlton *et al.* 2007); [2]First goat eradication: 1999 (Campbell and Donlan 2005); [3]Cows were introduced in 1985 but died due to competition with goats (Rico-Cerda pers. comm.); [4]Feral population eradicated in 2005, domesticated

Footnotes to Table 3 continued:

individuals removed alive in 2007 (Aguirre-Muñoz *et al.* 2009b); [5]First cat eradication attempt: 1980s by SEDUE; [6]Project conducted by UNAM (Rodríguez *et al.* 2006); [7]First eradication attempt, conducted by UNAM, failed (Rodríguez-Juárez *et al.* 2006). [8]Two cats were reintroduced and removed during 2002 (GECI 2003). [9]Project conducted by UNAM (Ramírez Ruiz and Ceballos-González 1996).

A= Donlan *et al*. 2000; B= Wood *et al*. 2002; C= Nogales *et al*. 2004; D= Knowlton *et al*. 2007; E= Aguirre-Muñoz *et al.* 2008; F= Aguirre-Muñoz *et al*. 2009a; G= Carrión *et al*. 2006; H= Aguirre-Muñoz *et al*. 2005; I= Aguirre-Muñoz *et* al. 2007; J= Luna-Mendoza *et al*. 2007; K= Tershy *et al*. 2002; L= Donlan *et* al. 2002; M= Arnaud-Franco *et al*. 2000; N= Rodríguez-Moreno *et al*. 2007; O= Howald *et al*. 2007; P= Samaniego-Herrera *et al*. 2009; Q= Rodríguez *et al*. 2006; R-= Samaniego-Herrera *et al*. 2010; S= Ramírez-Ruiz and Ceballos-González 1996; T= Donlan *et al.* 2003; U= Campbell and Donlan 2005; V= Álvarez-Romero *et al*. 2008; W= Arata *et al*. 2009; X= Ortiz-Alcaraz *et al.* 2009; Y=Sánchez-Pacheco and Tershy 2002; Z=GECI 2003; AA=Sánchez-Pacheco and Tershy 2000; AB=Hermosillo-Bueno pers. comm. 2010; AC= Sánchez-Pacheco and Tershy 2001; AD= Samaniego-Herrera *et al*. 2011.

the total restored area, especially in the past three years, when aerial broadcast methods were used, supported by on-board differential GPS, satellite imagery and telemetry (Samaniego-Herrera *et al.* 2011). The efficiency of helicopter aerial broadcast and hunting is illustrated by comparison with traditional ground-based methods. Ground-based traditional methods on 25 islands represent 26% of the total area, compared with aerial-based methods on five islands, but involving 74% of the total area.

Regardless of the methods used, the ultimate objective of an eradication project is restoration of ecosystems. Each project carefully evaluates the risks to non-target species and ensures that the long-term benefits are greater than the short term impacts that can derive from those activities.

Seabird restoration

When introduced species have extirpated populations of seabirds, action may be required to attract birds to return. There has been no natural recolonisation by six species on Asunción and San Roque Islands after 14 years without introduced predators. Attempts are now being made to attract the birds back using sound systems, decoys, and mirrors (Félix-Lizarraga *et al.* 2009), simultaneously with systematic and long term monitoring. These methods have been used successfully elsewhere (Kress 1978; Podolsky 1990; Gummer 2003)

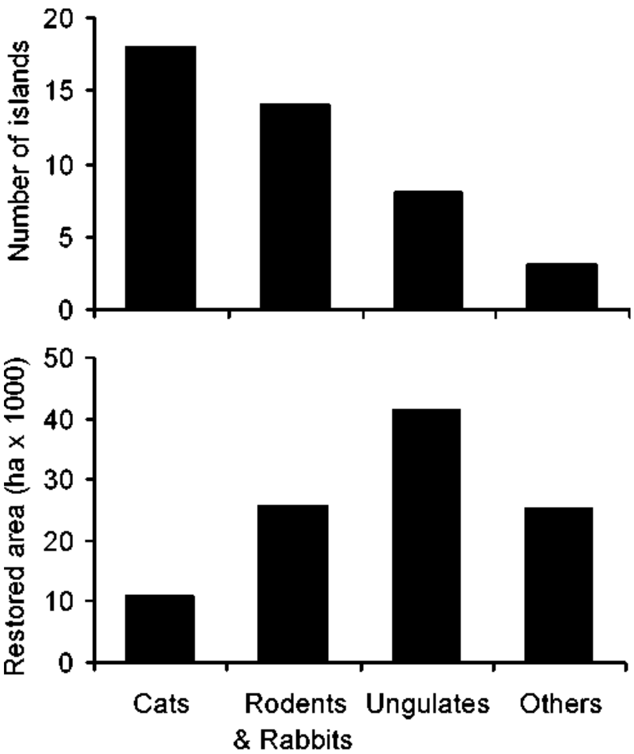

Fig. 3 Relationship between the number of islands with eradications and the restored island areas, by group of introduced species (see text and Table 3 for details).

ECOLOGICAL OUTCOMES OF ERADICATIONS

A lack of ecological information and the inherent richness of many island communities, pose challenges when evaluating, measuring and comparing the outcomes of eradications. Recovery may be documented for some species, but often the data are scarce and non systematic. Elsewhere, information has come from informal or anecdotal observations. Recently, there have been improvements in pre– and post–eradication monitoring, which allows more systematic evaluation of ecosystem recovery. The associated increase in cost remains a limiting factor.

Recovery of native species

On the Baja California Pacific islands, extirpated species such as Cassin's auklet (*Ptychroramphus aleuticus*), brown pelican (*Pelecanus occidentalis*) and Brandt's cormorant (*Phalacrocorax penicillatus*) have returned to breed (Wolf *et al.* 2006; Félix-Lizárraga *et al.* 2009). Four seabird species that have colonised islands represent new records; however, they could also have been extirpated long ago by cats and rats, before anyone recorded them. Vegetation also recovers. In the San Benito Archipelago, for example, two endemic species, the bush mallow (*Malva pacifica*) and the succulent live-forever (*Dudleya linearis*) are no longer at critical status after introduced rabbits were eradicated (Donlan *et al.* 2002, 2003).

In the Gulf of California, recolonisation by Craveri's murrelet (*Synthliboramphus craveri*) has been reported on San Pedro Mártir Island. Increased seabird reproductive success has also been documented, including a 60% increase in the nests of red-billed tropicbird (*Phaethon aethereus*) at Farallón de San Ignacio Island. There have also been new reports of plants, terrestrial birds, reptiles, and bats (Samaniego-Herrera *et al.* 2011, GECI unpublished data).

The bird attraction techniques used during the last two years are producing results. There are recorded interactions between elegant terns (*Thalasseus elegans*) and Heermann's gulls (*Larus heermanni*) and the attraction systems. During recent seasons these included placing nests with eggs among the decoys (Félix-Lizárraga *et al.* 2009).

The Socorro dove (*Zenaida graysoni*), endemic to Socorro Island, has been declared extinct in the wild. Merino sheep introduced to the island in the 1800s changed vegetative cover and structure. Later, cats and house mice were introduced. In combination, these introduced species are implicated in the extinction or endangerment of the endemic elf owl (*Micrathene whitneyi graysoni*), Socorro mockingbird (*Mimodes graysoni*), and Socorro dove. However, doves have been successfully breeding in zoos since 1987. The restoration process for reintroduction of Socorro doves to their native habitat is now under way.

Case study: Guadalupe Island

Guadalupe is a 24,171 ha volcanic island 250 km off the Baja California Peninsula (Fig. 1), being one of the most biodiverse and unique islands in the Pacific. The island has been the habitat for 223 species of vascular plants (17.5% endemic), eight species of seabirds (one extinct), eight species of endemic terrestrial birds (five extinct) and three species of pinnipeds. Its surrounding marine environment is also unique and diverse.

During the 19th and 20th centuries, 46 species of plants and eight species of mammals were introduced to the island; four of the mammals became feral (Moran 1996). Overgrazing by goats decreased forest coverage from 3850 ha to 85 ha (Rodríguez-Malagón 2006), desert scrub was decreased from 10,550 ha to 800 ha (Oberbauer 2005), and some vegetation communities completely disappeared. Invasive plants spread throughout the island. Feral cats were probably responsible for the extinction of six of the nine species of endemic birds and reduced populations of other birds and invertebrates. The hunting of pinnipeds during the 18th and 19th centuries almost destroyed populations of the northern elephant seal (*Mirounga angustirostris*) and Guadalupe fur seal (*Arctocephalus townsendi*) (Hanna 1925).

The eradication of goats from Guadalupe Island in 2003-2006 provided the first step towards restoration of the native vegetation, with spectacular responses by some native plants. Seedlings of endemic trees, which were absent in 2003, appeared, and by 2009 included the endemic cypress (*Cupresssus guadalupensis guadalupensis*), pines (*Pinus radiata var. binata*), palms (*Brahea edulis*) and native oaks (*Quercus tomentella*). Species of plants believed extinct have reappeared, including the western tansymustard (*Descurainia pinnata*), coyote tobacco (*Nicotiana attenuata*), dense false gilia (*Allophyllum gilioides*), Guadalupe savroy (*Satureja palmeri*), redflower currant (*Ribes sanguineum*), bruckbush (*Ceanothus crassifolius*) and common woolly sunflower (*Eriophyllum lanatum* var. *grandiflorum*) (Junak *et al.* 2005; Luna-Mendoza *et al.* 2007; W. Henry pers. comm.; J. Hernández-Montoya pers. comm.). The eradication of feral dogs in 2007 has helped to protect birds and pinnipeds from predation. Invasive mammals remaining on the island are cats and mice (Table 1). To prevent more extinctions, cats have been controlled around seabird nesting areas since 2003. The eradication of cats and mice poses a major challenge because of Guadalupe's size and complexity. Conservación de Islas, a Mexican NGO, is working with Federal Government agencies to assess the best options for the eradication of these mammals.

Guadalupe Island is now a Biosphere Reserve. Environmental education and social work has been undertaken with the local community to demonstrate how conservation actions help to improve quality of life. Future advances in restoration of this island should be of national and international significance.

DISCUSSION

Public policies and government involvement

There has been growing cross-institutional collaboration for island management, especially between agencies of the Federal Government and Conservación de Islas. SEMAR has provided invaluable and sustained logistic support, transportation and accommodation. Beyond a regulatory role, the Ministry of Environment (SEMARNAT), through the Wildlife General Direction (DGVS) has facilitated documentation and permitting. The National Institute of Ecology (INE) has supported restoration work with significant economic resources, especially for Guadalupe Island, the Marías and Revillagigedo Archipelagos. CONANP plays a key role in the implementation of eradication programs, and along with the US Fish and Wildlife Service and the National Commission for the Knowledge and Use of Biodiversity (CONABIO), managed significant economic resources in 2008-2009 for the restoration of Mexican islands.

Government involvement is now taking a step forward. Island restoration and conservation is now a national priority to preserve the country's natural heritage. In 2010, CONABIO published the "National Strategy on Invasive Species: Prevention, Control and Eradication", a document which highlights the priority tasks for the future. Furthermore, INE, SEGOB, CONANP and Conservación de Islas are integrating the "National Strategy on Island Conservation and Sustainable Development", which will complement with the one on invasive species (Karina Santos del Prado pers. comm.).

Challenges for the restoration of Mexican islands

Given the level of institutional support now being provided, the eradication of all introduced mammals from Mexican islands is a strategic goal that could be achieved by 2025. There are at least 41 islands with 832 populations of 12 species of introduced mammals, with rodents, cats, and goats being the most widespread. The greatest challenges are provided on bigger islands with complex terrain and ecosystems, the presence of native mammals, and interaction with human activities. One such example is Cedros Island (34,933 ha) with six species of introduced mammals, 12 endemic species (including five mammals), and a human population of 4500 inhabitants. Another challenge is the implementation of new techniques such as hunting methods, toxins, and viruses which may currently be illegal in Mexico. Success will also require the retention of skilled operators and specialised scientists, the development of new lines of research, and an appropriate legal framework.

Information is now being collected on introduction pathways, distribution of invasive species, and actions required to mitigate their effects through prevention, control and eradication. The advances outlined in this review represent unprecedented action to preserve and conserve the country's natural heritage. Eradication projects against other introduced species such as birds, reptile, amphibians, invertebrates, and plants have not yet been implemented, and the effects of such species remain unknown. There is an urgent need to create or update the inventories of invasive alien species on islands, and identify the ecological and economical impacts they have. There is also an urgent need to promote research on the ecology of invasions and methods for eradication. Interdisciplinary research is also essential to establish the relationship between the people and the uses and movements of invasive organisms. Preventing introductions of new invasive species as well as containing the spread of those already in the country both pose big challenges. Success will require the consolidation of the collaboration approach between government and academic institutions, NGOs, local communities and funders. Ecotourism must also be critically analysed and its regulations enforced.

Finally, if all Mexican islands are to be restored, a long-term and sustainable funding scheme, and appropriate legislation and policies will be needed to facilitate the control and eradication of invasive species.

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Return of endemic plant populations on Trindade Island, Brazil, with comments on the fauna

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Abstract Trindade (20º 30' S, 29º 20' W, 10 km^2) is an oceanic archipelago of volcanic origin, 1200km east of Vitória, the coastal capital of the State of Espírito Santo, South-eastern Brazil. The main island has lush terrestrial vegetation, with C. 130 species of vascular plants (10 of them endemics) and many endemic seabird species. Since the early 1700s, the forests which historically covered 85% of Trindade gradually dwindled to less than 5%, due to devastation by feral goats (*Capra hircus*), pigs (*Sus scrofa*), sheep (*Ovis aries*), and mice (*Mus musculus*). This change greatly reduced nesting opportunities for seabirds, especially the two endemic frigate birds. From 1965 to 1995, approximately 800 goats and thousands of mice remained, hindering the regeneration of vegetation. The eradication of goats, concluded in 2004, led to rapid revegetation of barren areas and the expansion of populations of endemic plant species *Psilotum triquetrum* Sw. f. *insularis*, *Achyrocline disjuncta., Peperomia beckeri*, and *Plantago trinitatis*, which were considered extinct or endangered in prior decades. The number of nesting *Sula leucogaster* nearly quadrupled following elimination of the goats and cats (*Felis catus*).

Keywords: Oceanic island, endemic species, invasive herbivorous vertebrates, eradication, vegetation recovery, *Bulbostylis*, *Colubrina*, *Cyperus*

INTRODUCTION

Trindade (20º 30' S, 29º 20' W, 10 km^2) is an oceanic archipelago of volcanic origin, 1200km east of Vitória, the coastal capital of the State of Espírito Santo, south-eastern Brazil (Fig. 1). The nearest islands to Trindade are Martin Vaz (50 km E), Ascensión (2130 km NE) and Saint Helena (2550 km ENE). Only the main island, Trindade, harbours significant terrestrial vegetation with more than 130 species of vascular plants. Many seabird species occur; some of them believed to be endemic. Among the 100 plus species of arthropods, the most conspicuous is the land crab, *Gecarcinus lagostoma*, which is also common on Ascensión (Pain *et al*. 2000). Scanty surveys of marine habitats have revealed relatively rich faunas, with several endemic species of fish and molluscs (Murphy 1915; Miranda-Ribeiro 1919; Carvalho 1950; Breure and Coelho 1976; Leal and Bouchet 1991).

The last volcanic activity on Trindade occurred in the Holocene (Almeida 2006), when eruptions around 30,000 years b.p. buried the forests in volcanic ash. These forests are now seen as recently exposed fossilised and preserved wood (Alves *et al*. 2003). In the late 17th Century, ship captains reported that Trindade was almost entirely covered with forest. Our mapping of fallen and buried tree trunks indicated that *Colubrina glandulosa* was the predominant species in these forests (Alves 1998).

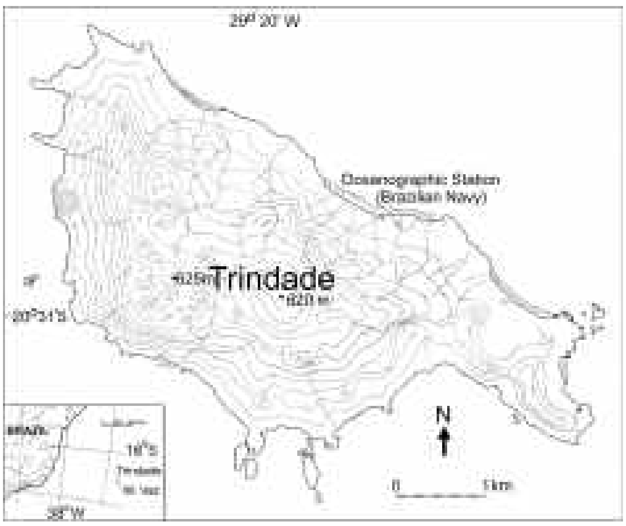

Fig. 1 Trindade Island, also showing the location of Martin Vaz Island.

Recorded invasions by vertebrates (Table 1) began in 1700, when Sir Edmund Halley introduced the first goats (*Capra hircus*), pigs (*Sus scrofa*) and guinea fowl (*Numida meleagris*) to the island (Copeland 1882; Thrower 1981). Between 1781 and 1782, the island was colonised for a year and two months by a 150-man English garrison under the command of commodore George Johnstone (Ribeiro 1951). Between 1785 and 1797 a new occupation by 200 Portuguese took place (Brito 1877; Azevedo 1898; Ribeiro 1951). During these occupations, the forests were overexploited, and the remaining trees were reported as dead, yet standing (Knight 1892).

All through the 19th and most of the 20th Century, introduced and invasive animals such as goats, pigs, sheep, cats (*Felis catus*), guinea fowl, and mice (*Mus musculus*) left on the island by fishermen and shipwrecks, prevented vegetation from recovering and exerted continued pressure on the terrestrial ecosystem. Populations of feral herbivorous, domestic mammals have now affected the terrestrial biota for more than three centuries.

In 1916, a research expedition to Trindade from the National Museum, Brazil, concluded that the introduced mammals were causing erosion and damage to the flora and fauna. Since 1957, the Brazilian Navy has had a permanent Oceanographic Post on Trindade, usually manned by 35 personnel. The Navy promoted sporadic efforts to eradicate introduced mammals throughout this period, but ironically introduced donkeys (*Equus asinus*) in order to pull cargo rafts from ships (Ribeiro 1951; Mayer 1981). Feral sheep, pigs, and donkeys, which were regularly hunted for food by the garrison, were eliminated by 1965 (Alves 1998).

Our field survey in 1994 (Alves 1998) revealed several hundred feral goats and the Navy ordered staff to eradicate them. The rugged mountainous terrain of Trindade posed many difficulties and about 200 goats were dispatched by 2002 by traditional ground hunting and, on one occasion, by helicopter. The Navy intensified the effort by sending

Table 1 Recorded vertebrate invasions on Trindade island.

Species	Period	References	Observation
Goat *Capra hircus*	1700–2005	Copeland 1882; Thrower 1981	Introduced by Edmund Halley, eradicated by Navy.
Pig *Sus scrofa*	1700–1965	Copeland 1882; Thrower 1981	Introduced by Edmund Halley, eradicated by Navy.
Guinea fowl *Numida meleagris*	1700– late 1980s	Copeland 1882; Thrower 1981	Feral population possibly reintroduced several times, eradicated by Navy
Sheep *Ovis aries*	1781–1965	Bücherl 1959	300 up to 1950s, eradicated by Navy
Domestic cat *Felis catus*	1783–1989	Delano 1817; Copeland 1882	Eradicated by Navy
Donkey *Equus asinus*	1916–1965	Ribeiro 1951	Introduced and eradicated by Navy
Cattle *Bos taurus*	1916	Ribeiro 1951	One pair, did not survive.
Canary *Serinus canaria*	?– 1985	Neves 1986	Small population, self–extinguished
Tropical house gecko *Hemidactylus mabouia*	2006– 2007–?	Bugoni & Welff–Neto 2008	15 indivíduals recorded.

Marine sniper hunters on several four month hunting missions, and thus the last 251 goats were shot by 2004 (Alves 2006). As the military personnel were on the island as a regular crew of the Oceanographic Post, the additional cost of this eradication effort was only that of the ammunition. In parallel, several thousand saplings, mainly of *Colubrina glandulosa*, were experimentally planted between 2000 and 2004. The feral cats, present at least since Amaro Delano's visit in 1803 (Knight 1892) decimated seabird populations, and were only eradicated by traditional ground hunting methods by the military in 1998 (Alves 1998).

TERRESTRIAL FLORA

Among the endemic vascular plants, *Bulbostylis nesiotis* and *Cyperus atlanticus* are common to the Trindade and Martin Vaz Archipelagos, while the remaining species (Table 2) grow, or grew, exclusively on Trindade.

Conservation results

Positive results of the goat eradication include the recovery of endemic plant populations. *Plantago trinitatis* was considered extinct until 1998, and began a slow recovery from the seed bank in the soil when the goat population began to decline. *Peperomia beckeri,* another endemic species known only from the type collection, was rediscovered in December 2009 and is now present as a few individuals. In 1994, the documented surviving population of *Achyrocline disjuncta* was of 13 individuals, with fewer than 50 individuals estimated for the entire island.

Areas kept barren by feral goats up to the 1990s are currently being colonised by herbaceous vegetation (Fig. 2). The chief pioneer species in this process are the endemic sedges *Cyperus atlanticus* and *Bulbostylis nesiotis*, followed by the fern *Pityrogramma calomelanos* (Alves and Martins 2004; Martins and Alves 2007).

Fig. 2 A ridge on Trindade Island in 1995, when hundreds of feral goats degraded the vegetation and impeded recovery (left) and the same area in 2009, five years after feral goat eradication. The herbaceous layer is composed mainly of the two endemic sedges *Cyperus atlanticus* and *Bulbostylis nesiotis*, and the widespread fern *Pityrogramma calomelanos*.

Table 2 Conservation status of plant taxa endemic to Trindade Archipelago.

Taxon	Discovered / described	Status
Asplenium beckeri	1965/1969	Extinct?
Doryopteris campos-portoi	1965/1969	Relatively common in shaded places.
Thelypteris sp. (=*Dryopteris novaeana*	1965/1969	Relatively common associated to *Cyathea copelandii* forest.
Elaphoglossum beckeri	1965/1969	Extinct?
Cyathea copelandii	1874/1882	Relatively common above 400 m a.s.l.
Polypodium trinidadense	1965/1969	Relatively common on exposed hilltops.
Psilotum triquetrum Sw. f. *insularis*	1965/1969	Considered extinct until 2000, currently expanding rapidly (ca 100 individuals).
Achyrocline disjuncta	1876/1885	Fewer than 20 individuals in 1990s, expanding rapidly (>100 individuals).
Peperomia beckeri	1965/1998	Considered extinct until 2009, recently recollected and in cultivation. Field survey in progress.
Plantago trinitatis	1965/1974	Twelve when discovered in 1965, after goat-eradication. expanding rapidly (ca 800 individuals on the tallest peaks).
Bulbostylis nesiotis	1876/1885	Common on Trindade and Martin Vaz; on Trindade now spreading to all barren land with fine soil.
Cyperus atlanticus	1876/1885	Common on Trindade and Martin Vaz; on Trindade now spreading to all barren land with fine soil.

Since 2004, the freshwater streams on Trindade have increased in volume and number; we found four new streams on the eastern flanks alone and observed that the total volume of water is about twice that of the late 1990s.

Several hundred saplings of about 60 non-native tree species were sent to Trindade and planted there without our knowledge, although luckily most of them were planted close to the barracks. Some of these saplings displayed a strange form of allelopathy, killing the native endemic *Cyperus atlanticus* within the reach of their root systems, and left a barren halo around their trunks. This is especially true for *Syzygium cumini* (Fig. 3), the halos of which are perceptible even on Google Earth satellite imagery. We recommend the substitution of the non-native allelopathic trees by native species.

EXTANT TERRESTRIAL FAUNA

Insects

The beetle, *Liagonum beckeri,* is almost certainly the world's most extreme example of narrow endemism. The population is restricted to a wet rock of <1 m^2, inside a deep ravine. About 20 individuals of this beetle are visible at any time. In 1959, it was discovered there by the late Professor Johann Becker, entomologist of the National Museum in Rio de Janeiro, and was described by Jeannel (1961). In 1994-95, we spent two months searching the entire Island, but only found the population on the very same spot as the Becker population. The beetles run around only on those parts of the rock that are covered with a green algal biofilm. The population was last revisited in 2002 and many individuals were observed.

Fig. 3 Allelopathic halos of dead *Cyperus atlanticus* within the reach of root systems of the mistakenly introduced tree *Syzygium cumini*.

Birds

The Trindade petrel *(Pterodroma arminjoniana*) is known to breed on Trindade, Round Island (Mauritius), and North Keeling Island (Australia, Cocos Archipelago) in the Indian Ocean. Luigi (1995) found no breeding pairs on Martin Vaz. It may also have nested on a coastal island in Espírito Santo, Brazil (Neves *et al*. 2006). There is no evidence to suggest that there is genetic exchange between the Australian and extralimital populations (Anonymous 2010). Unlike the frigates, this species nests and breeds on cliff ledges and in fissures, and does not depend on tree nesting. The petrel was listed as critically endangered (Neves *et al*. 2006, Silveira and Straube 2008) for Brazil, and the IUCN (2004) listed it as Vulnerable (D2). We have observed a gradual increase in the Trindade population coincident with the cat and goat eradication effort.

The boobies, *Sula sula* and *S. leucogaster,* have undergone a gradual global decline although both species are listed as "Least Concern" (Birdlife International 2009a, 2009b). They are not considered threatened in Brazil (not listed by Silveira and Straube 2008). Both are ground-nesting, and their populations on Trindade were under constant pressure from feral goats, which not only trampled their nests, but were recorded eating the eggs (Sergeant Ruy Barreto pers. comm.). Colonies of *S. sula* were recorded on Trindade up to the late 1960s, became very rare on the island by the 1990s, and no nests have been recorded since. On the other hand, the number of nesting *S. leucogaster* multiplied exponentially following cat and goat eradication, and currently covers four times the original territory.

Two critically endangered frigate bird subspecies *Fregata ariel trinitatis* and *F. minor nicolli* are endemic to Trindade and Martin Vaz Archipelagos (Silveira and Straube 2008). Even though frigate birds are believed to nest exclusively on trees, Martin Vaz has only herbaceous vegetation. On Trindade, no nest has been recorded since 1975 (Silveira and Straube 2008). During a visit in June 2009, we photographed a single pair of frigate birds soaring over the island (species not determined).

Reptiles

Between 2006 and 2007, a small and geographically restricted population ot the tropical house gecko (*Hemidactylus mabouia*) was recorded on Trindade, feeding mainly on exotic insects (Bugoni and Welff-Neto 2008). Whether the gecko is still present is uncertain, due to it´s nocturnal habits, but it has not been recorded since 2007.

Mammals

The only invasive vertebrate species now present on Trindade is the house mouse. The population is estimated to be in the order of hundreds of thousands of individuals. An assessment of their spatial distribution and seasonality on the island is being conducted. A detailed analysis of the house mouse's role in the Trindade food web is also pending. Preliminary observations indicate that mice consume most of the seeds produced on land, thus retarding vegetation recovery. They have been observed picking seeds of the endemic sedges *Cyperus atlanticus*, *Bulbostylis nesiotis*, and those of *Colubrina glandulosa.* The mice have also been seen foraging on eggs from seabird nests on the ground. Since the goat eradication was confirmed in 2004, and the island's vegetation shows clear signs of recovery, the invasive mice are the only significant setback to vegetation regeneration. Invasive rodents can also significantly change marine rocky intertidal communities (Kurie *et al*. 2008) and suppress terrestrial invertebrates (Van Aarde *et al*. 1996). A year–long rodent population survey has been started in February 2010, to help plan an eradication using methods proven effective on other islands (viz. Samaniego-Herrera *et al*. 2009a, 2009b). As proven by preliminary field trapping, land crabs are likely to pose a major difficulty by consuming large quantities of the bait (Fig. 4).

Fig. 4 The land crabs (*Gecarcinus lagostoma*) consumed most of the bait during preliminary field trapping in February 2010.

DISCUSSION

The order of eradication of invasive mammals was not considered during the goat hunting campaign. The feral sheep, pigs and donkeys had already been eradicated decades prior to our intervention, during which the goats were perceived as the largest threat. It could well be that the presence of goats, which kept the vegetation from recovering, may have facilitated the eradication of feral cats by ground hunting. It is also probable that by eliminating the goats first, we have helped the mouse population to increase. However, no hard data are available on these matters.

Due to its remote location, efficient management, and especially to the lack of economic exploitation, the recovery of terrestrial ecosystems on Trindade Island has begun with astonishing speed. The eradication of feral goats took a decade, but it was achieved without allocation of substantial resources – the salaries of the military personnel would be paid anyway and, considering the environmental benefits, the cost of ammunition was insignificant. Furthermore, the eradication represented an excellent training opportunity for the snipers.

Future introductions of any non-native species to Trindade should be subjected to prior evaluations by several specialists of different areas, in order to minimise potential impacts on the natural ecosystems. In the case of non-native fruit trees, for example, the benefits of their introduction must be weighed against the potential risks of their becoming new invasive species. The adoption of simple and preliminary biosecurity measures by the Navy would greatly benefit the Trindade Island biota, especially considering that without effective biosecurity measures, the upcoming eradication of house mice could easily be followed by a new invasion.

Provided with the right information, the Brazilian Navy has proven to be very efficient and conservation-minded, and we recommend that it should remain the sole administrator of Trindade Island. We consider that the adoption of simple biosecurity measures can benefit the environmental recovery of Trindade more effectively than the bureaucratic inclusion of the Island in the Brazilian National System of Protected Areas (SNUC).

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A targeted approach to multi-species control and eradication of escaped garden and ecosystem modifying weeds on Motuopao Island, Northland, New Zealand

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Abstract Motuopao Island (30 ha), on the north western tip of the North Island, New Zealand, comprises 118 m high basaltic stacks covered in sand. It holds substantial breeding populations of black-winged petrel (*Pterodroma nigripennis*) and common diving petrel (*Pelecanoides urinatrix*). The island was a manned lighthouse station between 1879 and 1940 and was grazed by sheep (*Ovis aries*). In 1989, the Department of Conservation eradicated Pacific rat (*Rattus exulans*) and in 1997 it commenced control of weeds including Madeira vine (*Anredera cordifolia*), smilax (*Asparagus asparagoides*) and *Gladiolus cardinalis*, wallflower (*Cheiranthus cheiri*), and tree mallow (*Lavatera arborea*). Weed control was initiated to control the spread of the remaining garden plants and to remove ecosystem impacting plants before restoration of native habitats. Annual visits were not controlling or eradicating the remaining targeted species, and since 2005 two visits have been undertaken during April-May and October-November each year. Removal rates in the 0.2 ha gridded area of Madeira vine has declined from 1.25 kg per person hour in November 2006 to 0.66 kg per person hour in April 2009. Smilax has been recorded within 3.5 ha, and the amount removed has fallen from 5.5 kg per site in November 2005 to 0.36 kg per site in April 2009, as older plants and then seedlings have been removed. Tree mallow has been controlled over the accessible regions to prevent seeding, and the two former areas that had a mallow canopy have started to revert to grassland. Gladioli and wallflower are near elimination.

Keywords: Madeira vine, Tree mallow, *Anredera cordifolia*, *Asparagus asparagoides*, *Gadiolus cardinalis*, *Cheiranthus cheiri*, *Lavatera arborea*

INTRODUCTION

Motuopao Island (30 ha, 34° 28"S, 172° 38'E) lies 500 m west of Cape Maria van Diemen on the north-western tip of the North Island, New Zealand. The island comprises two 118 m tall basaltic stacks covered in sand and a saddle valley that runs in an east and west direction (Fig.1). The mean temperature is 15.5°C, and the mean annual rainfall 1058 mm (Tomlinson and Sansom 1994). The island is frequently windswept by strong south-westerlies and access to the island is difficult due to the swells and 5-10 knot currents in the channel between Motuopao Island and Cape Maria van Diemen.

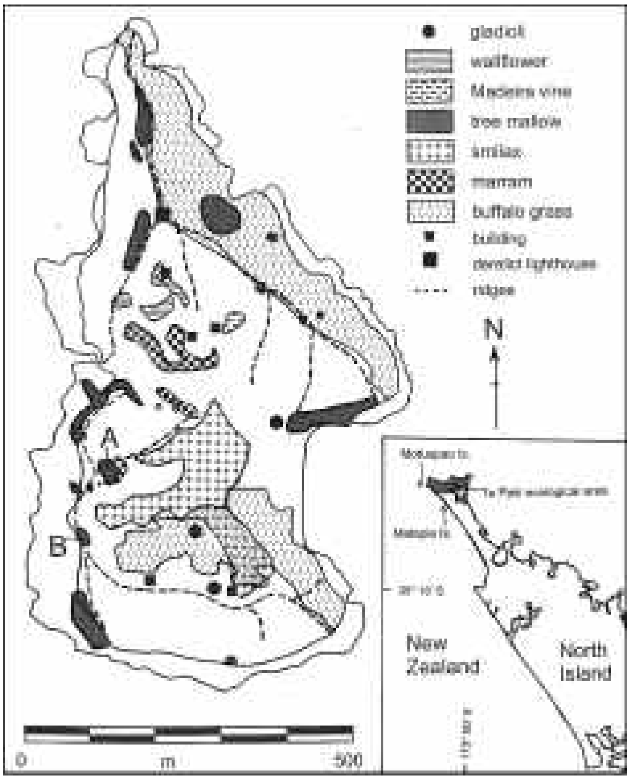

Fig. 1 Location of weed regions on Motuopao Island, Northland, New Zealand. A and B are closed canopy tree mallow sites.

Motuopao Island was used in pre-European times as a Maori fishing camp. It was originally covered in coastal forest but this was destroyed before European times (Forester 1993). In 1879 a wooden lighthouse was erected on the northern stack and housing for three families was established below the lighthouse. The central valley and lower slopes of Motuopao were substantially devegetated in 1902, and sand movement led to the decision in 1921 to replace the lower houses with two new ones on the southern slope (Shirley 1985). The island remained a lighthouse station until 1940 when the light was dismantled and taken to Cape Reinga. Photographs taken during the time of occupation show that the central part of the island was mobile sand and that the slopes had similar amounts of vegetation cover as seen today (Shirley 1985; Beaglehole 2006). Sheep skeletons suggest some grazing took place (Forester 1993).

Surveys between September 1988 and February 1990 found that Motuopao Island had six species of breeding petrels including common diving petrels (*Pelecanoides urinatrix*), grey-faced petrels (*Pterodroma macroptera*), black-winged petrels (*Pterodroma nigripennis*), white-faced storm petrels (*Pelagodroma marina*), sooty shearwaters (*Puffinus griseus*), and fluttering shearwaters (*Puffinus gavia*). It also had three species of skinks: shore skink (*Oligosoma smithi*), moco skink (*O. moco*) and Suter's skink (*O. suteri*) and there were Pacific geckos (*Hoplodactylus pacificus*) on the northern stack.

The island was visited in 1981, 1983, and four times between September 1988 and February 1990 by botanists who compiled a list of 133 vascular plants and a vegetation map (Forester 1993). This list included 30 rare and threatened plants and other relict plants from human occupation including red-throated gladiolus (*Gadiolus* *cardinalis*), jonquil (*Narcissus tazetta*), wallflower (*Cheiranthes cheiri*), pink flowered stock (*Matthiola incana*), Indian mustard (*Brassica juncea*), rape (*B. napus*), wild cabbage (*B. oleracea*) and potato (*Solanum tuberosum*). Marram (*Ammophila arenaria*) grassland was the dominant vegetation in 3% of the central valley, and buffalo grass (*Stenotaphrum secundatum*) was the predominant vegetation covering 60% of the island (Forester 1993). Two native communities, one dominated by flax (*Phormium tenax*) and the other dominated by taupata (*Coprosma repens*) and ice plant (*Disphyma australe*), were present on southern faces, and rock faces and comprised C. 20% and C. 2% of the cover on Motuopao Island, respectively.

A few patches of tree mallow (*Lavatera arborea*) was scattered across the island (Forester 1993). Smilax (*Asparagus asparagoides*) was seen in December 1983 but not in the later visits in September – October 1988 and February 1990 (Forester 1993). Madeira vine (*Anredera cordifolia*) was discovered in 1997 (T. McCluggage and A. Booth pers. comm.) and was confined to one site which appears to have been near out-buildings used until 1921. If left unchecked vegetative spread could threaten habitats used by flax snail (*Placostylus a. ambagiosus*), Matapia gecko (*Hoplodactylus* aff. *pacificus "Matapia"*) and mawhai (*Sicyos australis*). Monitoring and control caught boxthorn (*Lycium ferocissimum*) and tree lupin (*Lupinus arboreus*) before they gained any foothold on the island (Forester 1993). Pampas (*Cortaderia selloana*) and apple of Sodom (*Solanum linnaeanum*) were not present in 1990 (Forester 1993), but have appeared and subsequently been removed.

Pacific rat (*Rattus exulans*) were detected in 1988 and were eradicated by February 1990 (McKenzie 1993). Pacific rat were found to be eating the threatened endemic flax snail. Lizard restoration commenced in 1998 with the release of robust skink (*Oligosoma alani*) and geckos from neighbouring Matapia Island, 21.5 km to the south.

The garden snails (*Helix aspersa*), introduced during the lighthouse period, were very abundant in the grasslands but were scarce or absent in the broadleaf shrublands where they were considered a potential problem to the recovery of native plants and flax snail; the latter through food competition (Parrish and Sherley 1993).

Native revegetation and the spread of weeds on the island were limited due to a lack of nearby seed sources, and *in situ* seed-distributing lizards and birds (Pierce and Parrish 1993). Silvereyes (*Zosterops lateralis*) were seasonally common when taupata fruited in January 1992 after the Pacific rat eradication, and introduced song thrush (*Turdus philomelos*) and blackbirds (*T. merula*) were rare.

Current strategic planning is aimed at ensuring the survival of the threatened species on the island and the introduction of other threatened endemics within the Te Paki ecological area (Fig.1). The plan necessitates an island-wide native vegetation restoration programme but decisions on the type and extent of plant and animal communities have not been made. The first priority was to remove weeds that would prevent revegetation.

The objectives of the current weed programme on Motuopao Island are: (1) to exterminate potential pest weeds *Gladiolus*, wallflower, smilax and Madeira vine by the removal of all seeds, tubers and bulbs by 2016; (2) to control tree mallow at accessible sites by 2016; and (3) to investigate ways of restoring native vegetation without causing sand erosion.

This paper reports on the progress towards these objectives, the changes in management that have taken place to eradicate and control weeds and the various measures put in place to monitor progress of weeds that are in the control phase.

METHODS

Weed control approach

Motuopao Island is generally only accessible by helicopter and was visited irregularly between 1997 and 2003, annually between 2003 and 2005 and twice a year in April-May and October-November since 2006. In 2006, the frequently of visits was altered to ensure plants were controlled before viable seed was set.

Madeira vine

Madeira vine was initially controlled in 1997 and 1999, by digging and bagging tubers and vegetative material and placing Tordon G2 granules where tubers were detected. The bags were placed on sand below the infected site to heat treat the tubers (R. Renwick, pers comm.). In 2003-04, glyphosate was painted onto the cambium of scraped stems to try to poison the tubers. After two years, assessment of coverage indicated that this is was not successful, and control reverted to digging initially from the middle of the infestation. In April 2007, the site was divided into 41, 5x10 m weed control plots and the priority changed to controlling the perimeter. At each subsequent visit priority was given to previously worked plots before starting on a new plot. The time spent on each weed control plot was recorded. The weight of vegetation removed and time spent in each plot was recorded. From 2001, all Madeira vine was double bagged and weighed before removal from the island to a landfill. From 2007, specific boots and clothing were used, left on site, and double bagged for removal and cleaning at the end of the trip.

Smilax

Smilax was controlled from April 2005 by removing the vegetation and tubers from spatially separated infestations of different size (sites). A numbered post and GPS coordinates demarked sites for further follow-up. Full delimitation was not carried out before November 2007 when the island was also swept using staff with hip chain walking 15 m apart and the time take for each phase of the operation was recorded. In all years, the vegetation and tubers were double-bagged and weighed before they were removed from Motuopao Island.

Gladioli

Five sites with gladioli were dug and all visible bulbs were removed from 2005 to 2007.

Wallflower

Wallflower plants were pulled and the ripe seed collected and removed from Motuopao Island from May 2006. After 2008, the site was grid searched and the coordinates of the remaining positive sites recorded for further follow-up.

Tree mallow

Tree mallow control started in April 2005 by pulling plants in the central valley. In May 2006, pulling and cutting was investigated to see if it would control the largest areas where canopy dominance of tree mallow occurred (Fig. 1 sites A and B). Photo points were established at each site. In November 2006, the extent of the canopy dominated sites was defined with poles, and the petrel burrows within two 5x5 m plots at site A were mapped to see how control affected petrel use of the area. In 2006, active burrows were defined as those with petrels in them or that had recently cleared sand entrances and tunnels, and inactive burrows as those with entrances had mallow debris and seed head within them. In 2009, active burrows were defined visually as in 2006, and verified during three consecutive nights using sticks placed at the tunnel entrances, which were knocked down by entering or exiting petrels. From November 2007, all accessible tree mallow plants on Motuopao Island were pulled or cut and two litres of seed heads were taken for future planting at a mainland controlled site to assess seed viability. Subsequent visits were timed to cut or pull plants before they added to the seed bank, and any flower heads were bagged and removed. In April 2009, after substantial control of key sites had been established, the numbers of plants that were cut or pulled at all sites were counted.

RESULTS

Madeira vine

The Madeira vine infestation was C. 850 m² confined to one valley in flax and was surrounded by vines of the threatened native curcurbid, mawhai (*Sicyos australis*). A third, a quarter, and two thirds of the infested area was cleared in April 1997, April 1999 and January 2001, respectively.

An April 1999, inspection of the bags of vine and tubers placed on open sand in 1997 found that wind and sun had destroyed the exposed upper surfaces of most bags. Seventy five percent of the contents of bags were dead, and in those bags holding water the tubers had rotted. Regrowth from remaining live tubers had been suppressed by grazing of the new shoots by garden snails. In 2001, the plastic-lined depression dug to hold water and the remaining live 1997 tubers and the 1999 tubers and vegetation was inspected and some tubers still were viable. These tubers were removed from Motuopao Island in 2001. In 2005 and 2006, visual assessments of sites where the cambium of Madeira vine had been scraped and painted with glyphosate found substantial vine was still growing. In October 2007, all of the perimeter plots were dug and 153 kg of material was removed. In April 2009, all 41 plots were finally controlled during a single trip and 417 kg of material was removed. The weight of material collected per person hour of search time is now declining (Fig. 2) and we expect this decline to continue.

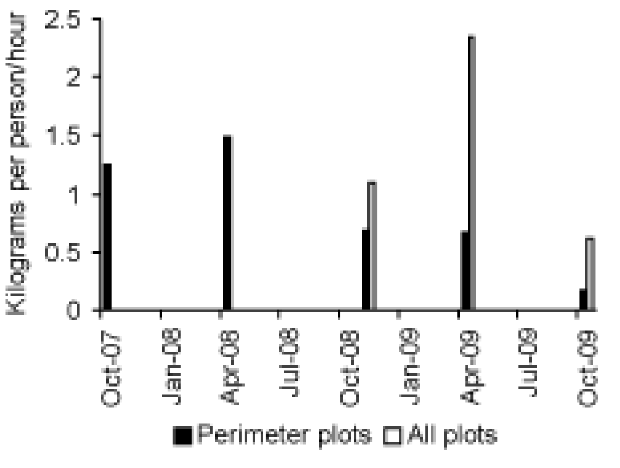

Fig. 2 Madeira vine removed with search effort (kg per person hour) on Motuopao Island, New Zealand.

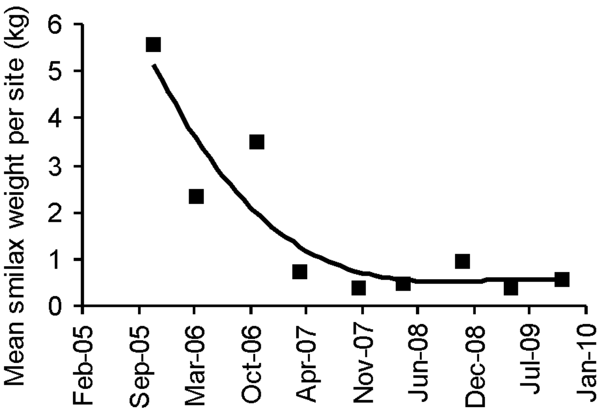

Fig. 3 Changes in weight of smilax (kg per site) removed from Motuopao Island, New Zealand.

Smilax

Smilax was distributed at point sources in the southern central valley and at one point near the lighthouse. Initially some point sources were old plants with many tubers. In November 2005 111 kg, and April 2006 94 kg of smilax was removed from Motuopao Island. Subsequently, most point sources have been seedlings and the average amount removed per site has fallen from 5.5 kg per site in November 2005, to 0.36 kg per site in April 2009 (Fig. 3), within a region of 3.1 ha. Smilax re-growth, or grown from seed, has been detected at only 16 of 78 sites, and new sites are the result of seed bank germination.

Gladioli

Gladioli were removed from within 120 m² in the vicinity of the southern house site and near the historic lighthouse stores landing site (Fig. 1). In November 2008, plants were still being detected at three sites and all soil to 50 cm deep was removed from the island. No plants were seen in the April or November 2009 site or grid searches.

Wallflower

Wallflower was known from the vicinity of the remains of the northern house site (Fig. 1); 206 plants were pulled in November 2006, and 27 plants were pulled in November 2007 and all potentially viable seed was bagged and removed. No plants were found in April and November 2009.

Tree Mallow

Opportunistic cutting and pulling of tree mallow prior to May 2006 had almost completely removed patches in the central valley. Pulling and cutting part of the two closed canopy tree mallow sites (75 m² and 300 m², Fig 1 sites A and B) in May 2006 resulted in substantially lower seeding densities and regrowth as measured using photo points. In November 2006, both sites were cut completely, and most of the material was removed and the rest was left within the sites. The controlled sites started to grow over with exotic grass (*Bromus* spp.) and canopy dominance ceased in 2007. Subsequently, both sites were controlled before seed was set, and control in autumn has reduced the numbers of mature plants requiring attention the following spring. In April 2009, 15,086 and 14,981 plants, and in October 2009, 1940 and 2000 plants, were removed from sites A and B, respectively. Seed collected in November 2007 was stored dry in the light and remained viable in December 2009.

Two 5x5 m plots established at site A (Fig. 1) to assess petrel use during control found the number of active burrows in 2006 was 11 and 11, and in 2009 was 17 and 9 in plots one and two, respectively. In 2009 plot one had only diving petrel present and in both plots the location of active burrows has changed as the site has reverted to grasses.

Ongoing control has now been extended to cover other accessible parts of the island. In April 2009, 68,358 plants and in October 2009, 12,367 plants at were pulled or cut at 11 sites.

DISCUSSION

Motuopao Island is an important habitat for threatened plant, snail, lizard and seabird populations and is potentially an important site for the translocation of other endemic plants and insects that are threatened in the Te Paki region. The types of habitat that are present now are substantially modified but representative of habitats on the mainland.

The current weed programme has targeted garden plants from the lighthouse days that if left unchecked would slowly become widespread. The programme was not making inroads into the remaining weeds in 2005 because the trips were too infrequent and were not directed at the right times of the year to stop flowering and seed dispersal. The redirecting of the programme in 2007 to a mid autumn and mid spring time frame, with adequate staffing to accomplish the tasks set for that trip, has been a key to the ongoing success. Gladioli and wallflower appear to be close to eradication. This has only been gained by removing soil in the gladioli sites and ensuring that we stopped the seed production of wallflower. As control was established, the C. 40 hours of time used per visit to manage these two species has been redirected to other weeds.

The main emphasis now is the eradication of Madeira vine and smilax and the control of tree mallow; species that pose a greater risk to native biota. These three species are tackled together each trip with the aim of optimising the impact but ensuring we first control tree mallow flowering, then search for and control all smilax, and then put the remaining time into systematically reducing the Madeira vine infestation. Successful control of flowering of tree mallow has meant that the cut or pulled material does not need to be removed from Motuopao Island. Grid searching for smilax has also been able to be used to reassure us that other species are not appearing on Motuopao Island.

The control of Madeira vine has been achieved by removing the core tubers in the infestation, griding the area and then concentrating on the perimeter and then working systematically over the entire area. In this way we maximised the weight of vine that was removed from the island by helicopter each trip. The reduction in growth and detection has been shown in the increase in person time per kilogram removed. Madeira vine does not set seed in New Zealand and is generally associated with old house sites and rubbish dumps. It may have been planted historically at some sites as an herbal laxative (Tony McCluggage pers. comm.). The main risk is vegetative spread by people controlling weeds, so footwear and clothing quarantine actions have been put in place while it is exterminated. We have not found new sites on the island and the clothing and footwear quarantine methods used here have been effective.

Importance of tree mallow removal

Tree mallow has been identified as a threat to native species and restoration on other historically highly modified New Zealand islands, including on Mana, Tahaka and Motunau Islands (Bannock 1998). Tree mallow is a problem on islands that have been modified, grazed and then are taken over rapidly by seabirds (Rippey *et al.* 2002).

Tree mallow became a problem on Craigleith Is, Scotland, when myxomatosis eradicated the rabbits that were potential grazers on seedlings. Gull (*Larus* spp.) and puffin (*Fratercula arctica*) populations expanded increasing nutrient loadings, and frost frequencies declined. Puffin burrow entrances were good establishment sites for mallow as they were moist, fertile and had few plant competitors, especially dense grass swards. Eventually mallow canopy hindered access, and the puffin population declined from 28,000 to 12,100 burrows (van der Wal *et al.* 2008).

Information from Motunau Island, Canterbury, New Zealand suggests that tree mallow could alter the habitat and current distribution of the petrel populations on Motuopao Island and lead to reduced sites for storm petrels (Beach *et al.* 1997), and lizard populations (Bannock 1998).

Tree mallow took over the Shoalwater islands, Western Australia within 30 years (Rippey *et al.* 2002). It reduced plant biodiversity and restricted the tern breeding sites. Tree mallow's mass death in droughts was likely to expose the resulting bare ground to erosion (Rippey *et al.* 2002). Similar erosion concerns are present on Motuopao Island.

Tree mallow covered half of Mud Island (30 ha) in Port Philip, Victoria, in 1994 and cutting and pulling reduced the infestation to a few seedlings in seven years (Rippey *et al.* 2002). This indicates that at least seven years of seeding prevention may be necessary on Motuopao Island to have a substantial impact on the seed bank. However, the time frame to substantial control on Motuopao Island could be altered by petrel exposure of open germination habitat around burrow entrances.

Pacific rat were removed from Motuopao Island to improve the status of the flax snails and lizards (Parrish and Pierce 1993). No consideration was given to the ecological changes that would result from eradication on weed and plant populations. Tree mallow was widespread in 1997 on Motuopao Island (R. Renwick pers. comm.). Pacific rats may have been exerting pressure on the seed bank of mallow until 1990 (Pierce and Parrish 1993; Rippey *et al.* 2002) and we are fortunate it had only reached canopy dominance at a few small sites by 2006. If tree mallow was left unchecked it could form a canopy on much of Motuopao Island within 30 years (Rippey *et al.* 2002).

Future programme

Weed monitoring will be ongoing as most of the plants that have been controlled have tubers or, in the case of tree mallow, long-lived surface propagating seeds (Okusanya 1979; Rippey *et al*. 2002) and the seed bank is constantly being buried and re-exposed by borrowing seabirds. Three species of brassicas were present on Motuopao Island in 1988-90 (Forester 1993) and one re-appeared and was controlled in 2005. Other species like pampas grass were establishing on the neighbouring mainland in 1993 and have appeared on Motuopao Island and will remain a threat (Forester 1993).

Tree mallow is well established on some of the cliff sections of the island and will need to be controlled at key wind funnel sites as the seeds could be distributed from there by storm force winds to some parts of the island we are controlling. Cutting has occurred at all the accessible sites and chemical methods need investigation.

There are two other major ecosystem impacting weeds: marram and buffalo grass present on the island and have not had substantial control. Flax has been transplanted into the marram areas and this is likely to continue. No decisions have been made on what will be done with buffalo grass. Buffalo grass now covers 30% of the island and has stabilised major areas of sand.

If the tree mallow control programme is successful the operational plan will be revised to take into account all of the likely consequences of removal or modification of further plant/weed communities, and assess which habitats will need to be maintained to retain the current biota and released threatened species. Currently, regular monitoring is restricted to ants and the two translocated lizards (A. Booth pers comm.). There are also some petrel burrow maps and plots that will be useful for modelling changes with various restoration actions. Some research has assessed the composition of the native snail fauna on Motuopao Island (Parrish and Sherley 1993) and threatened invertebrates in the Te Paki ecological area (Goulstone *et al.* 1993; O. Ball and P. Whaley pers comm.). The status of the native plants on Motuopao Island that were potentially suppressed by Pacific rat (Campbell and Atkinson 2002) needs to be reassessed. A full habitat-correlated invertebrate assessment, and petrel burrow and lizard reassessment is needed to ensure that the future weed programme is fully integrated into habitat restoration and management.

CONCLUSIONS

The majority of the escaped garden plants from the lighthouse period have been eradicated from Motuopao Island. All known gladioli sources remained active until the soil to 50 cm deep was removed. Wallflower was only controlled when visits were altered to capture plants before seed production. The major point sources of smilax have been removed and seedlings are still being detected and removed. A single Madeira vine site is now under control. Removal rates from the grid-covered site are indicating that we can expect to eradicate this plant within 3-5 years. Tree mallow is being controlled to prevent addition to the seed bank at accessible sites. Twice yearly visits in autumn and spring are able to cut and pull the plants on all accessible sites. Revegetation of weed sites is taking place with introduced and native species. Two potential ecosystem modifying weeds, marram and buffalo grass, are yet to be tackled. Their removal needs to take place after consideration of the success of tree mallow control, the impact of vegetation changes on petrel breeding sites, and the vegetation restoration pathways we want to encourage in Motuopao Island.

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The ground-based eradication of Norway rats (*Rattus norvegicus*) from the Isle of Canna, Inner Hebrides, Scotland

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Abstract: Seabird populations, particularly European shags (*Phalacrocorax aristotelis)* and Manx shearwater (*Puffinus puffinus)*, on the Isle of Canna have been in decline for several years. Norway rats (*Rattus norvegicus*) were identified as the most likely factor. The Canna Seabird Recovery Project, developed as a result of this information, was a three year project incorporating the ground-based eradication of Norway rats (Phase I) followed by long-term monitoring, quarantine and contingency for rodents (Phase II) and continued long-term monitoring of the seabird populations (Phase III). The National Trust for Scotland (NTS) contracted Wildlife Management International Limited to direct the first and second phases of the project (August 2005-June 2008), with the assistance of NTS staff and volunteers. Techniques, problems, non-target species, (particularly wood mice (*Apodemus sylvaticus)* and white-tailed eagles (*Haliaeetus albicilla)*) issues, solutions and results of the operation on the permanently inhabited island are covered. Bait stations with cereal-based wax blocks containing diphacinone at 0.005% w/w were established on a fifty to one hundred metre grid over the island. Some offshore islets harboured rats, although some of the more sheer stacks did not. Interference with bait stations by non-target species was moderate to high, and bait stations required extra strengthening or protection to prevent damage or disturbance by cattle (*Bos taurus*), ponies (*Equus caballus*), sheep (*Ovis aries*), rabbits (*Oryctolagus cuniculus*) and hooded crows (*Corvus corone cornix*). Monitoring confirmed the successful eradication of rats from the Isle of Canna in June 2008. This provides another example of the effectiveness of ground-based rodent eradication techniques and provides an opportunity to restore the seabirds.

Keywords: Wood mouse, white-tailed eagle, diphacinone, monitoring, quarantine and contingency

INTRODUCTION

The Isle of Canna is located off the west coast of Scotland in the Inner Hebrides (6°30'W, 57°03'N), and consists of two semi-connected main islands; Canna (1126 ha) and Sanday (191 ha), and several small offshore stacks and islets (Fig. 1). The Highland Ringing Group, which has monitored the seabird colonies of the Isle of Canna for over 40 years had recorded that seabird populations (in particular razorbills *(Alca torda)*, European shags *(Phalacrocorax aristotelis)* and Manx shearwater *(Puffinus puffinus)*) had been declining since the early 1990s (Swann 2002). Brown (Norway) rats (*Rattus norvegicus*) were identified as the most likely factor influencing this decline, from the observed increased predation on eggs and chicks (Swann 2002). Rats are known to have devastating effects on seabird populations, causing extinctions of birds on numerous islands throughout the world (Moors and Atkinson 1984; Atkinson 1985; Jones *et al.* 2008). Many islands have been successfully cleared of rats (Thomas and Taylor 2002; Howald *et al.* 2007) with a subsequent increase in bird populations (Towns and Broome 2003; Jones *et al.* 2008).

The National Trust for Scotland (NTS) commissioned a feasibility study into the potential for the eradication of rats from Canna (Bell and Bell 2004), based on an earlier proposal (Patterson 2003). The Canna Steering Group, a partnership of NTS, Edinburgh Zoo, Scottish Natural Heritage (SNH), the Joint Nature Conservation Committee (JNCC) and Royal Society for the Protection of Birds (RSPB), decided that eradication of rats using a ground-based eradication technique was to proceed. Wildlife Management International Limited (WMIL) won the tender to direct the eradication with the assistance of NTS volunteers and staff. The three-phase Canna Seabird Recovery Project (Phase I eradication of Norway rats; Phase II monitoring for surviving rats and implementation of quarantine and contingency procedures; Phase III long-term monitoring of seabirds) began in August 2005 (Bell *et al.* 2006). Complete details of the project are available on the project website (www.ntsseabirds.org.uk).

STUDY AREA AND METHODS

Canna and Sanday are naturally joined at low tide and are now linked by a road bridge. Canna is approximately 8 km long, east to west, and 2 km across at its widest point.

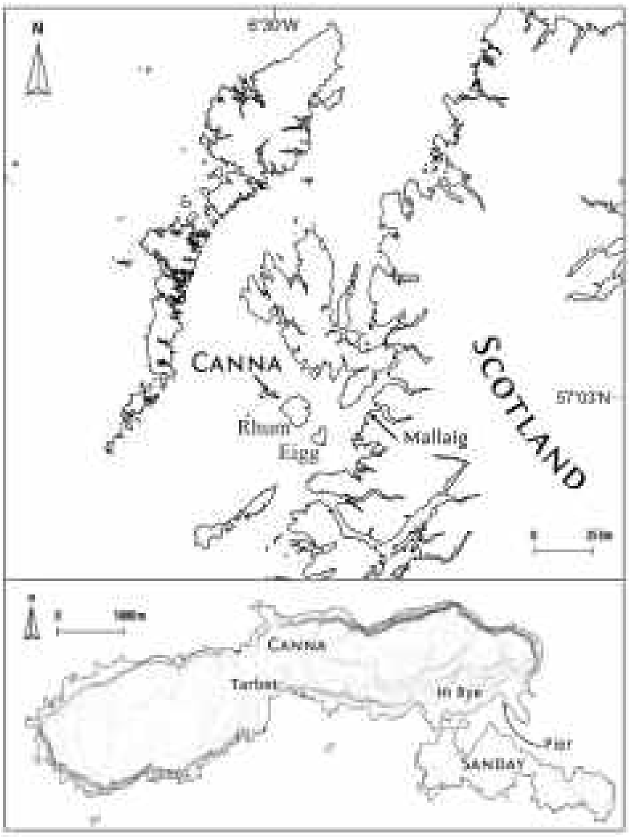

Fig. 1 Location of Isle of Canna, Inner Hebrides, Scotland.

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With the exception of the in-bye land close to the farmhouse and buildings, Canna rises steeply from beach platforms on all sides to a rolling plateau with the highest point at 210 m above sea level. Sanday is approximately 3 km long, east to west, and just over 1 km at its widest point. Sanday reaches 59 m a.s.l., but is similar to Canna with steep coastal cliffs, beach platforms and a low rolling plateau.

Owned and managed by NTS, the Isle of Canna has 15 permanent residents who maintain the farm and crofts, or manage the tourism ventures. The islands (i.e. both Canna and Sanday) are popular with visitors interested in the seabirds, raptors, flora and history. There are several houses, crofts, farm buildings, churches, a lighthouse and school. Canna has a jetty and is regularly serviced by the Caledonian MacBrayne ferry from Mallaig.

The islands are covered by maritime heath, coastal pasture and heather moorland, apart from Tarbert and in-bye areas, which are improved pasture. There are also small areas of mixed woodland adjacent to the in-bye land. The island is grazed by domestic livestock including horses (*Equus caballus*), sheep (*Ovis aries*), and cattle (*Bos taurus*); three feral goats (*Capra hircus*), and rabbits (*Oryctolagus cuniculus*). Canna also has three other small mammals; the pygmy shrew (*Sorex minutus*), house mouse (*Mus musculus*) and wood mouse (*Apodemus sylvaticus*). Hedgehogs (*Erinaceus europaeus)* are also present. A small number of domestic cats and dogs are kept by the residents.

The Isle of Canna (excluding all the in-bye land) was designated a Site of Special Scientific Interest (SSSI) in 1987 for its biological and geological features. Following this, Canna was also designated in 1997 as a Special Protected Area (as part of the European Union NATURA 2000 network of important bird sites) for its internationally important concentrations of breeding seabird species. Shags, razorbills, kittiwakes (*Rissa tridactyla*), puffins (*Fratercula arctica*), guillemots (*Uria aalge*), black guillemots (*Cepphus grylle*) and fulmars (*Fulmarus glacialis*) all breed on the island. However, many of these species are now in decline (Swann 2001, 2002). Historically, Manx shearwaters were also recorded to breed on the island, but have declined to almost zero (Swann 2002). Two pairs of white-tailed eagles (*Haliaeetus albicilla*) and a pair of golden eagle (*Aquila chrysaetos*) breed on the Isle of Canna, along with buzzards (*Buteo buteo*), peregrines (*Falco peregrinus*) and kestrels (*Falco tinnunculus*).

It is not known when rats became established on Canna; but this is likely to have occurred more than two hundred years ago, either as an accidental introduction with supplies or from an early shipwreck. Only the Norway rat is known from the island and previous surveys recorded them in all habitat types (Patterson and Brough 1999; Patterson and Lloyd 2000; Patterson and Quinn 2001; Patterson 2003). Distribution, however, was not uniform with the highest densities occurring around the in-bye land, the shoreline and coastal slopes; a pattern common to most island situations where rats are dependent on foraging for food in inter-tidal zones and at seabird colonies. Other notable rat presence on the island was generally related to farming activity, watercourses and rabbit habitat (Patterson 2003).

The eradication option adopted for this project was a ground-based poison programme using protective bait stations to reduce risk to non-target species, particularly the white-tailed eagle and other raptor populations. The programme ran from 25 August 2005 to June 2008 and included bait station establishment, capture of wood mice, poisoning, monitoring, quarantine and contingency, and a final check and rat-free declaration (Table 1). Each operational task was undertaken and completed as follows:

Bait station grid

The bait station grid was established between 3 September and 27 October 2005. Bait stations were made from 750 mm lengths of 100 mm diameter corrugated plastic drainage pipe, pegged to the ground with wire "legs" to prevent movement by wind and/or stabilised with rocks or other material to reduce interference by sheep, cattle, and ponies. Additional wires pushed through both entrances reduced the entrance size to exclude smaller non-target species such as rabbits, hooded crows (*Corvus corone cornix*) and gulls and to help secure the station to the ground. Both entrances were raised slightly off the ground to deter entry by insects.

Bait was placed in the centre of the station through a small access hole cut in the top which was covered by an additional short clip-on section of pipe as a lid. "Crow clips" (a piece of wire across the station), as used during the Lundy Island rat eradication (Bell 2004), were also used to prevent crows from removing lids to access bait.

Bait stations were placed on a 50-metre grid on the coastal slopes and cliffs, the in-bye area and on Sanday (Fig. 2). On the higher plateau areas on Canna, stations were placed more widely at 100 m (Fig. 2). All areas, except steep or sheer cliffs with no vegetation had bait stations. All offshore rock stacks and islets had bait stations, as did areas with sizeable vegetation below steep cliffs with difficult access. Ropes and a boat were used to access these areas.

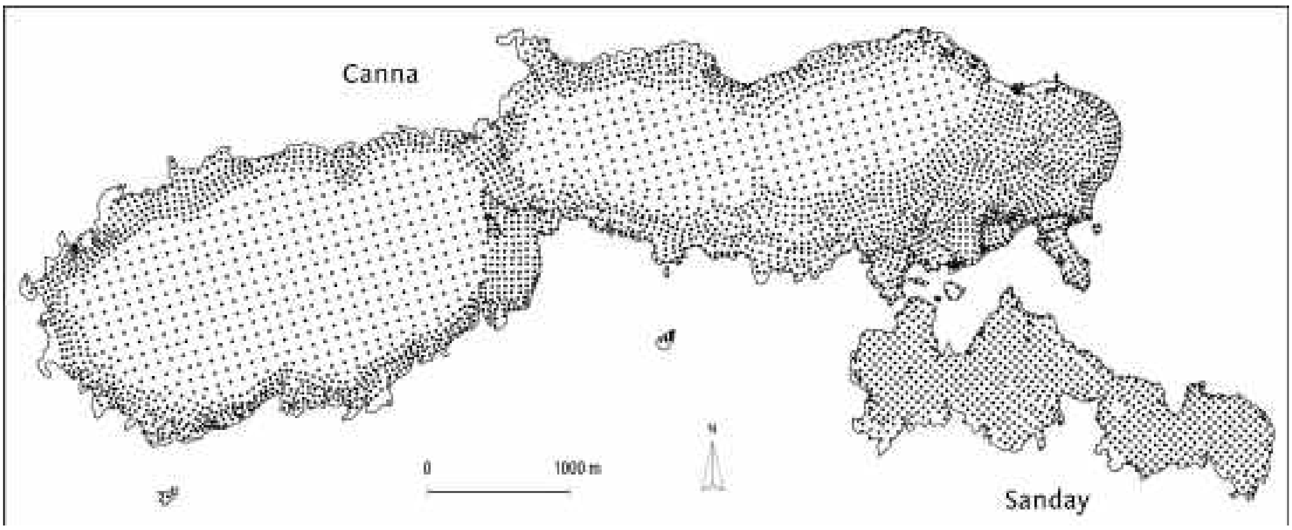

Fig. 2 Bait station grid on Canna and Sanday (bait station positions are marked by a black dot).

In outdoor areas, each station was marked with a cane and flagging tape to ensure visibility in thicker vegetation or during foggy conditions.

Tube bait stations were also positioned along the Beach Road and at the pier. Philproof and/or Protecta lockable stations were used inside all buildings.

Canna has numerous archaeological sites. WMIL, NTS and Historic Scotland worked together to identify important sites and,whenever possible, bait stations were placed outside recognisable structures (e.g., stone walls and remnant houses). If this was not possible, one or two stations were positioned in the best possible way to minimise disturbance or damage to the site. These sites were identified on maps for the field team and access to all archaeological sites was limited to work purposes only.

The entire grid of 4388 stations was positioned before being individually numbered and mapped using GIS (Manifold). Since Norway rats are reputedly neophobic and can be wary of new items placed in their environment, the grid was left for a period of two to six weeks to allow the rat population time to become familiar with it.

Capture of wood mice

It is thought that the race of wood mouse found on Canna is distinct, possibly a sub-species of the wood mouse found on the Scottish mainland (Berry *et al* 1967; Lloyd 2000; Patterson 2003) as it is larger and more golden than the mainland population (Patterson 2003). Eradication programmes can have an associated risk that non-target species will be poisoned either by direct consumption or through secondary poisoning. Principle preventative methods can include the design of the bait station, but the Canna wood mouse was small enough to gain direct access to the bait as well as being at risk from secondary poisoning by eating invertebrates that have eaten the bait. The spacing of the bait station grid meant the chance of accidentally eradicating the Canna wood mouse was unlikely (due to their small home range), but it was decided that a small, but viable 'assurance' population would be live-captured and held as two captive sub-populations at Edinburgh Zoo and the Highland Wildlife Park for the duration of the baiting period.

The translocation of wood mice was undertaken by staff from The Royal Zoological Society of Scotland during the bait station establishment period from 8 September to 3 November 2005. Longworth live traps were deployed in a range of habitat types and locations over Canna and Sanday. Traps were run for three nights at each site before being moved to alternative sites. Traps containing bedding materials were baited with grain and invertebrates and checked every four hours. A total of 158 wood mice were captured. All individuals were maintained and transported in North Kent Plastic MB1 laboratory rat cages which minimised handling, and transferred without loss to Edinburgh Zoo and the Highland Wildlife Park.

Poisoning

First generation rodenticides were chosen for the eradication campaign to minimise the risk of secondary poisoning, particularly to birds. The main toxicant used was a 28 g, cereal-based wax block bait with 0.005% active ingredient diphacinone (Ditrac, manufactured by Bell Laboratories). The other rodenticide used was also a 28 g cereal-based wax block bait but with 0.005% active ingredient bromadiolone (Contrac, also manufactured by Bell Laboratories). However, only three blocks of Contrac bait were deployed at one location. Both types of bait were dyed blue (or green/blue), which makes them less attractive to birds.

The bait was delivered to Canna on 28 October 2005 and was transported to depots around the island by tractor and trailer and/or all terrain vehicle (ATV) and trailer.

The poisoning programme (Phase I) commenced on 1 November 2005 and continued through to 6 March 2006.

Baits were present in each station throughout the poisoning programme and replaced as required, when eaten by rats or non-target species and/or damaged by weather. Ten bait blocks were available in each bait station for most of the programme but this was reduced to three blocks when rat activity waned towards the end of the poisoning phase (6 January 2006). By mid February these bait blocks were wired into the stations to ensure missing baits were being taken by surviving rats rather than non-targets as crows and cattle sometimes shook the stations to displace and consume the bait.

The majority of stations (n = 4229) were checked and serviced every three to six days (November and December 2005) or every 15 to 20 days (January to March 2006). However, difficult to access bait stations (n = 66) in tide, weather, rope or boat dependent areas) were loaded with thirty blocks per station and checked whenever possible. Permanent bait stations (n = 93) established in the farmyard and buildings around the island were regularly inspected and maintained as required with ten blocks into each station.

To present the data on bait take gained from these varied bait station checks we grouped the data into 12 periods or checks (mean (±SEM) = 9.6 ± 1.8 days between checks, range 3-22 days) shown as days from baiting (Figs. 3 and 4).

Towards the end of the poisoning phase (15 February - 6 March 2006), when isolated incidents of rat activity, such as teeth marks or droppings, were detected in a monitoring or bait station, an additional bait block was staked inside the entrance of an identified rat hole in the vicinity and/

Table 1 Timetable of activity on the Canna Seabird Recovery Project

Dates	Phase	Activity
27 August 2005		Team arrive on Canna
3 September to 27 October 2005		Bait station grid established
8 September to 3 November 2005	PHASE I	Capture of wood mice
28 October 2005		Poison arrives
1 November 2005 to 6 March 2006		Poisoning operation
13 December 2005 to 27 March 2006		Intensive monitoring (M1)
28 March to 23 September 2006		Long-term monitoring (M2)
24 September to 19 December 2006		Intensive monitoring (M3)
20 December 2006 to 10 March 2008	PHASE II	Long-term monitoring (M4)
11 to 28 March 2008 2 to 9 May 2008		Final check and quarantine and contingency audit (M5)
7 June 2008		Declaration of rat-free status

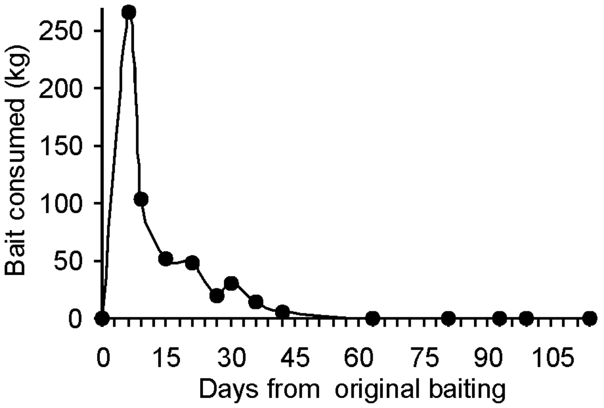

Fig. 3 Amount (kg) of bait consumed by rats at each bait check (marked by black dot) during the Norway rat eradication on the Isle of Canna, Inner Hebrides, Scotland.

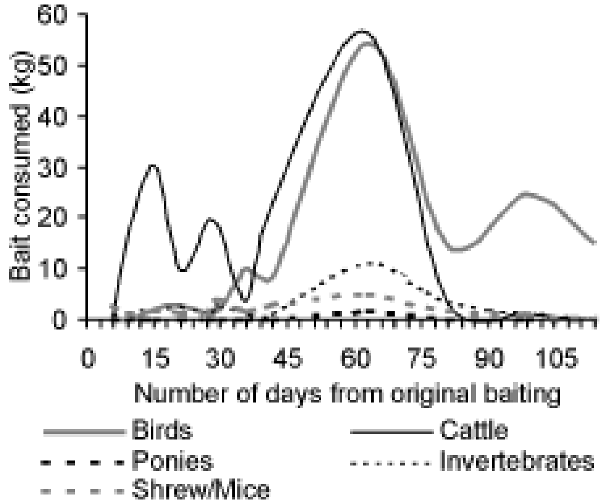

Fig. 4 Amount (kg) of bait consumed by non-target species at each bait check (marked by black dot) during the Norway rat eradication on the Isle of Canna, Inner Hebrides, Scotland.

or inside a purpose made ‘natural’ bait station such as a tunnel made from rocks. Both types of bait were used on these occasions, placed out of sight and reach of non-target species by covering entrances with rocks, vegetation or wire netting. These natural sites were marked with flagging tape, numbered and added to the bait station grid.

Bait take was recorded in field notebooks by bait station number and the species believed to have consumed or removed the bait. These data were entered into a data base and large-scale maps showing active stations were produced in real-time to enable the team to effectively monitor bait take activity and target any “hot spots”. All rat corpses found were collected and returned to base for safe disposal to reduce risk for non-target scavengers.

Monitoring

Five distinct periods of monitoring were undertaken as the project progressed (Table 1). Intensive monitoring (M1) using 5296 stations at 50 m spacing was carried out from 13 December to 27 March 2006 to detect rats surviving through the poisoning phase. This was followed by a six-month period of long-term monitoring (M2), from 28 March to 23 September 2006. A second intensive monitoring period (M3), utilising 7608 stations, was completed from 24 September to 18 December 2006 followed by a period of long-term monitoring (M4) using 801 stations from 20 December 2006 to 10 March 2008. These were established at high risk areas on the island; around the coastal seabird breeding sites, Beach Road, at the pier, around the farmyard, in out-buildings, in all properties and around the coast of Sanday (Bell *et al* 2006, 2007, Table 1). The final check (M5), using 1610 stations, was carried out between 11 and 28 March 2008. WMIL staff and NTS volunteers carried out the intensive and final checks and NTS staff maintained the long-term monthly monitoring over summer. There were two types of monitoring stations using rat attractive food items; one was secured to the ground by a wire and the other was secured inside a tube station. Both were individually numbered and any evidence of activity (i.e. teeth marks) was recorded in field notebooks by station number and the species believed to have consumed or marked the monitoring item.

Monitoring items such as soap, chocolate, chocolate wax, and candles (but most frequently chocolate wax) were placed inside and outside each station. Mud traps of mud smoothed out to detect rat foot prints were established on stock feeding sites, Tarbert Barn, Beach Road and at the pier. Checking for active rat burrows and rat runs, along with trapping at Tarbet Barn, was also undertaken.

Each monitoring site was checked regularly, either separately or together with the poisoning bait station grid. Both rat and non-target species sign found on detection devices was recorded and added to the database. If rat sign (usually tooth marks) was detected, an intensive poisoning (bait stations at 25-m) and trapping (snap traps) programme was established around the site.

RESULTS

Bait acceptance and take

Green/blue rat droppings appeared within five days of baiting and rats accounted for 540 kg of bait taken (estimated 3000-5000 rats). The bait take pattern was typical of other Norway rat eradication campaigns (Thomas and Taylor 2002). It was very high six days after original baiting (1st check) and dropped to a relatively low level 28 days after original baiting (5th check). A small increase was recorded at day 32 after the original baiting (6th check), but dropped away to a low level throughout the rest of the poison programme, reaching zero bait take on day 64 after the original baiting (9th check) (Fig. 3).

Throughout the poisoning phase, 62% of bait stations were visited by rats, with 50% active within the nine days of the original baiting. The low percentage of active stations shows that rats were not distributed evenly across the island nor were they in high numbers. This was reflected in bait take levels on the slopes. Ten percent of the bait stations had more than 12 blocks taken, and 3% had more than 21 baits taken by rats. On the plateau, 12 blocks were taken from 4% of the stations and 21 blocks from 1% of stations (Fig.5).

The coastal cliff areas, where breeding seabird colonies are established during summer, also had high bait take by rats, as did sites at Geugasgor, Lamasgor, Iolasgor, and the Nunnery where shag colonies are present during summer. There were few stations on the cliffs or slopes that had no bait take by rats (Fig. 5).

Bait take was also high on the offshore stacks and littoral areas of the main island accessed by boat (Fig. 5). Every bait station on the rock stacks had at least ten bait blocks taken, as did many of the shoreline stations on Canna.

The average number of blocks taken by rats was 8.06 (± 1.01) blocks per active station (n = 2732). The average number of blocks taken per station (n = 4388) was 4.4

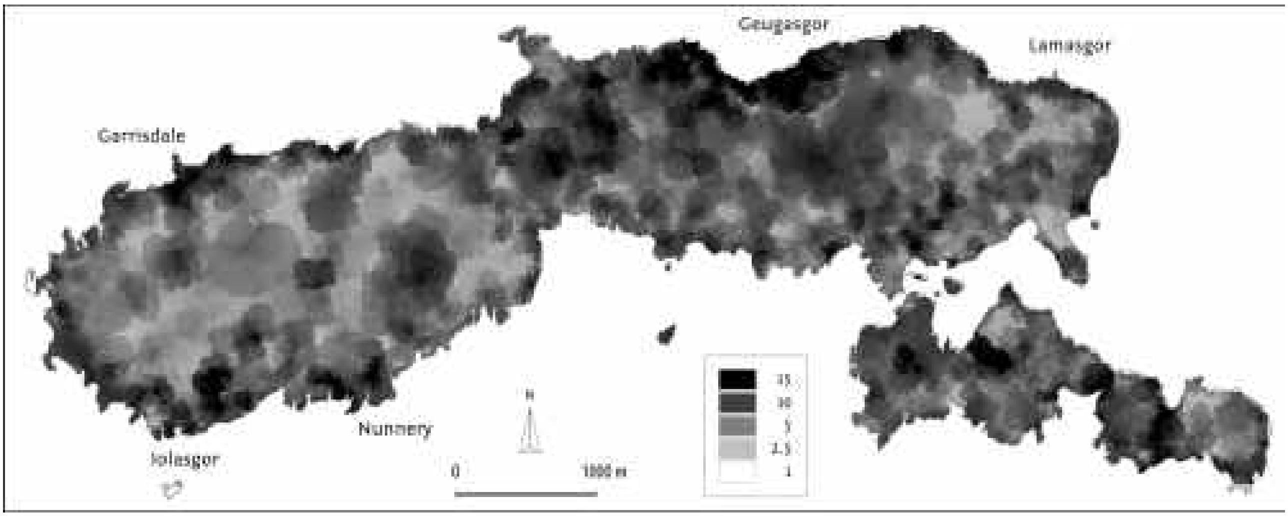

Fig. 5 Distribution of total bait take by rats (as bait blocks consumed per station) during the entire Norway rat eradication on the Isle of Canna, Inner Hebrides, Scotland.

(± 0.1). However, 38% of stations had no bait taken by rats and 54% had six or fewer blocks taken by rats.

There were low to moderate levels of interference by non-target species (Fig. 4). Cattle trampled bait stations, ate bait (<200 kg) and removed numbered tags. Ponies, sheep and goats had minor impact that was generally related to removing numbered tags or knocking over poles. Stock were not affected by the poison during the eradication. Crows (and other birds) ate moderate amounts of bait (<120 kg) and green regurgitation pellets were seen at known crow roost sites. No birds were found killed by poison during the eradication operation. Wood mice (< 15 kg), pygmy shrews (<2 kg) and insects and slugs (<25 kg) consumed small amounts of bait. Five wood mice and two pygmy shrew carcasses were found during the eradication operation. More than three tonnes of bait was lost to weather events, particularly on the coastal cliffs during storms.

Monitoring

Monitoring for rat presence continued island wide (Canna and Sanday) for two years after the end of the poisoning operation.

Three rats were detected on Canna during the four-month period when monitoring overlapped the end of the poison operation (last rat detected on 20/2/06). These were caught using traps and the alternative Contrac bait. No rats or sign were detected during monitoring after the end of the poisoning operation. Rat-free status for the Isle of Canna was declared in June 2008.

The wood mouse population on both Canna and Sanday recovered quickly after the eradication. Wood mice tooth marks were recorded at more than 75% of the monitoring points during sessions M1 to M5. Rabbits left tooth marks on devices at 23% of the monitoring points and pygmy shrew tooth marks were recorded at 17% of monitoring sites.

DISCUSSION

The success of the Isle of Canna rat eradication campaign shows that a well-planned, adequately resourced, well-executed programme, supported by the landowner and residents and directed by experienced operators can eradicate brown (Norway) rats from a large inhabited island using a ground-based poisoning technique.

Once set up, the island was cleared of rats within six weeks (42 days from original baiting; 9th check) with very few secondary and primary non-target species affected (and these mainly restricted to wood mice and pygmy shrews). Bait-take showed that the rat population was low to moderate and not evenly distributed across the island. High concentrations on the coastal slopes meant rats would have had an effect on nesting seabirds.

Problems encountered were few and mainly limited to weather and interference with bait and monitoring stations by non-target species. Although wood mice were recorded taking bait and a small number of losses did occur, the population quickly recovered in numbers and range after the removal of rats. Since the wood mouse population was recovering naturally there was no requirement to reintroduce individuals taken into captivity. Ten of the captive Canna wood mice have been used in an unrelated mark-recapture study and the remainder held at the Highland Wildlife Park as a permanent display.

There is no doubt that the eradication of Norway rats from Canna will benefit the recovery of breeding seabirds. Manx shearwaters were presumed to be extinct on the island, but a few individuals were still present (Swann 2008) and the first chick to be recorded on the island in ten years was found and banded in September 2006 (A. Ramsay, Caledonian Ornithological Services pers. comm.). This increases the possibility for successful recovery of the Manx shearwater population. There are also increases in productivity and/or numbers of puffins, razorbills and European shags (Swann 2008; Bob Swann pers. obs.). The seabird populations will continue to be monitored by the Highland Ringing Group.

With rats gone from Canna, it is important that they are never provided with an opportunity to re-establish on the island. As a permanently inhabited island, the greatest risks of rats reaching Canna comes from infested fishing boats mooring overnight, from equipment and food being brought to the island (via the Caledonian MacBrayne ferry or other vessels); and with visitors to the island. A rodent quarantine and contingency plan was developed which minimises the risk of rats being re-introduced, without being too onerous for island residents, ongoing projects, and visitor programmes (Bell *et al.* 2007, 2008; Bell and Garner-Richards 2006).

Bait stations have been established on the mainland (on Mallaig pier) and on the neighbouring islands of Rum, Eigg, and Muck which are not rat-free, but have some level of rodent control. Bait stations are maintained on the Caledonian MacBrayne ferry and landing-craft that service the island. Bait stations and trapping points have been established on Canna pier, Beach Road, farm buildings, tearooms, food storage areas, residents' homes, and guest accommodation. Rodent 'motels' (i.e. large wooden boxes that act as an attractive shelter and nesting area for rodents into which traps and monitoring items can be placed) have been placed in all high risk areas. All staff and residents on the island have been trained in quarantine methods, rodent sign and detection. One resident NTS staff member has been made responsible for enforcement of quarantine and implementation of any contingency action. Rodent-proof areas have been identified for unpacking suspicious or high-risk containers. All visitors to the island and boat owners mooring offshore are informed of the rat-free status of the island and are asked to be vigilant for rats and rat sign.

A contingency protocol was developed for Canna that details procedures for interviewing persons who report a rat sighting, inspecting the location of the sighting, determining if this is a likely rat event, establishing and maintaining monitoring, trapping and/or baiting grids, identifying tooth marks (or other sign) and reporting and recording all incidents (Bell and Garner-Richards 2006).

Rodents have now been successfully eradicated from islands ranging in size from 1 to 11,200 ha throughout the world. The successful eradication of rats from Ailsa Craig (100 ha; Zonfrillo 2001, 2002), Handa Island (Stoneman and Zonfrillo 2005), Ramsey Island (256 ha; Bell *et al.* 2000), Lundy Island (500 ha; Bell 2004) and now the Isle of Canna (1300 ha), demonstrates how ground-based poisoning operations can be effectively applied on islands around the UK and Europe. The success on Canna builds on the efforts of many projects that have gone before and lessons learnt will be invaluable for future eradication programmes, particularly those with important non-target species. It also shows that ground-based eradication techniques can be adapted for, and undertaken on, permanently inhabited islands of various sizes, and serves as a good example of the significant long-term benefits that can be achieved through short-term investment.

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Preliminary ecosystem response following invasive Norway rat eradication on Rat Island, Aleutian Islands, Alaska

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Abstract The Aleutian Islands, including many of the islands in Alaska Maritime National Wildlife Refuge, are among the most productive seabird breeding areas in North America, providing habitat for >10 million seabirds representing 26 species. Norway rats (*Rattus norvegicus*), accidentally introduced to several islands in the Aleutians, have had a negative impact on seabird populations. To reverse these effects, and recover seabird breeding habitat on Rat Island (2900ha) where burrowing seabirds have been extirpated, an eradication of Norway rats was attempted in September 2008. The project was undertaken by U.S. Fish and Wildlife Service partnered with The Nature Conservancy and Island Conservation, and was the first in Alaska to apply rodent bait aerially. No signs of rats were detected during a reconnaissance visit nine months after the operation when preliminary signs of positive responses were recorded, including notable breeding records of shorebirds and seabirds. Numerous bird carcasses, including glaucous-winged gulls and bald eagles, were found following bait application and toxicology analysis confirmed that most mortalities were direct or indirect effects of consumption of the baits. Nevertheless, with the exception of bald eagles, most bird populations surveyed increased in abundance so the impacts on non-target species are likely to be temporary. Monitoring on the island will continue for five years to further evaluate ecosystem and non-target species population recovery following rat removal.

Keywords: Invasive species, rat eradication, seabirds, *Rattus norvegicus*

INTRODUCTION

Introduced species are one of the top drivers of extinctions in island ecosystems worldwide. Island endemics are particularly vulnerable, as they often lack evolved behavioural responses to predators, or have restricted habitats or population sizes (Moors and Atkinson 1985; World Conservation Monitoring Center 1992). Increasingly, the removal of non-native predators is being used as a tool to prevent further loss of island biodiversity and restore native ecosystems to their original state. Introduced rodents are among the most detrimental mammals to island flora and fauna (Moors and Atkinson 1985) and, given their widespread colonisation and impact on native species, have been identified by land managers as key species for eradication.

The Aleutian Islands, including many of the islands in Alaska Maritime National Wildlife Refuge, are among the most productive seabird breeding areas in North America, providing habitat for 26 species of seabirds numbering more than 10 million individuals. Islands in the Aleutian Archipelago, however, have not been spared from the impacts of non-native species (Ebbert and Byrd 2002). Populations of ground nesting birds, and other native species in the Aleutians, have been depleted or, in some cases, entirely extirpated through predation by introduced species (Bailey 1993). Because of the high biodiversity values, the restoration of Aleutian Island ecosystems through the removal of invasive predators has been a long-standing management priority (Ebbert and Byrd 2002). For the past 50 years, restoration of Aleutian Island ecosystems has focused on removing introduced Arctic foxes (*Alopex lagopus*), resulting in dramatic population increases for 15-20 bird species (Gibson and Byrd 2008) and the de-listing of the endemic Aleutian cackling goose (*Branta hutchinsii leucopareia)* from the U.S. Endangered Species List. Additional species continue to be threatened by Norway rats (*Rattus norvegicus*) introduced to at least ten large islands in the archipelago.

Rat Island is thought to have been the first island in the Aleutians to be invaded by Norway rats when a Japanese ship went aground in the 1780s (Black 1983). Over the past two centuries, rats have caused extensive ecological damage by depleting breeding seabird and possibly land bird populations, and altering island plant and intertidal communities (Kurle *et al.* 2008; Croll *et al.* 2005). Arctic foxes were introduced to Rat Island by fur traders in the 1800s, but were removed in 1984 in the initial phase of native habitat restoration (Hanson *et al*. 1984) leaving Norway rats as the only remaining non-native mammal. The rats are a significant obstacle to further native habitat restoration.

The U.S. Fish and Wildlife Service, partnered with The Nature Conservancy, Alaska, and Island Conservation, to restore native biodiversity, including seabird breeding habitat, on Rat Island (2900ha) by removing introduced rats using an aerial application of cereal pellets containing 25ppm brodifacoum. Here we report generally on the aerial broadcast operations, in addition to biological surveys conducted before and after bait application to: 1) assess the potential impact to non-target species; and 2) document the recovery of native species following rat removal. We also provide preliminary results from target species monitoring to evaluate the effectiveness of the rodenticide bait in achieving rat removal.

METHODS

Island description

Rat Island (51°80' N, 178°30' E) is in the Rat Islands group in the central Aleutian Islands (Fig. 1). The 2900ha (7100acres) island has steep coastal cliffs around most of the coastline backed by rolling hills and plateaus rising to a small range of mountains with a maximum elevation of 400m. Rat Island is a designated Wilderness Area and has no inhabitants or infrastructure. The Aleutian climate is marine-influenced and is characterised by generally overcast skies and frequent, often-severe, storms driven by low-pressure systems and high winds (Rodionov *et al.* 2005). Rat Island is treeless and supports a subarctic maritime tundra ecosystem. The island has a diverse bird fauna including waterfowl, birds of prey, shorebirds, seabirds, and landbirds. Burrow-nesting seabirds appear to be absent and crevice-nesting species are rare, likely due to the impact of rats.

Pages 275-279 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

Eradication operation

Rats have been successfully eradicated from at least 330 islands worldwide, generally using an application of rodenticide bait to every potential rat territory on an island (Howald *et al.* 2007). The method used on Rat Island followed techniques used on large island eradications in New Zealand and elsewhere, but the details were adapted to suit the Aleutian environment (Towns and Broome 2003; Howald *et al.* 2007; Broome 2009). Cereal pellets (Brodifacoum 25W Conservation, Bell Laboratories, Madison, WI, EPA Registration # 56228-36) containing 25 ppm brodifacoum, a second generation anticoagulant, were applied twice from a specialised spreader bucket slung beneath a helicopter at a nominal sowing rate of 8.0 kg/ha (Buckelew *et al.* 2008). Bait was delivered during fall (September – October), when rats are relatively deprived of food by seasonal declines in resources and more likely to consume the pellets. Application was by flying low-altitude (*c.* 50m) parallel swaths over the entire land area and adjacent vegetated islets. A differential global positioning system was used to direct coverage across the island and ensure all individual rats were exposed to a sufficient quantity of bait. Directional deflectors were placed on the spreader buckets when applying bait to coastal and riparian areas to minimise the discharge of bait to marine and freshwater habitats. Bait was hand laid inside these aerial exclusion zones to ensure comprehensive coverage.

Biological surveys

Minimising the impacts to non-target species was a consideration in the eradication design; however, it was recognised that there might be mortality of some individual birds. Common birds were surveyed to document the recovery of native species following rat removal and to assess the impacts to non-target species in 2007 and 2008. The surveys were repeated in June 2009, nine months after the bait application. The bird population abundance indices obtained from these surveys were then compared. Additional surveys of marine mammals, vegetation, and intertidal biota are not discussed in this report (Buckelew *et al.* 2009). As much as possible, a Before-After design with replication (using the island as the inferential space) was used since logistical constraints, in most cases, precluded the use of sampling island replicates as a control. The survey methods used include point count surveys, strip transects, nearshore boat surveys, and incidental observations (Buckelew *et al.* 2007a). Values were tested using a two-sample or paired t-test ($\alpha = 0.05$)

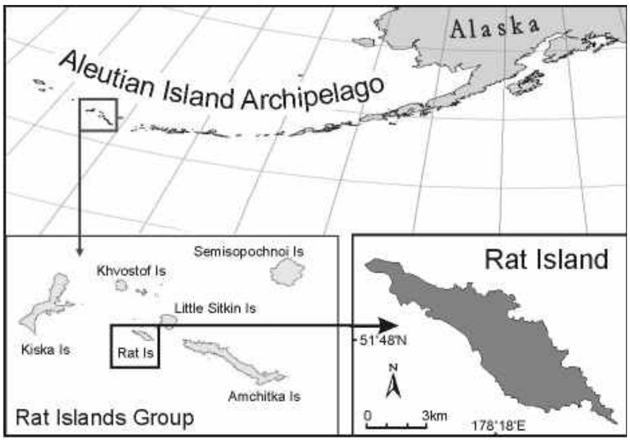

Fig. 1 The location of Rat Island in the Rat Islands group, central Aleutians Island unit of the Alaska Maritime National Wildlife Refuge.

Eradication confirmation monitoring

Following common practice, final determination of the eradication outcome will not be determined until two years after bait application, allowing time for any surviving rats to repopulate to detectable levels. Preliminary rat detection monitoring from May-June 2009 used transects of trap stations and chew devices. Thirty-one transect lines were placed along the coast and in riparian habitats. A transect line consisted of ten trap stations and ten peanut butter-flavoured wax chew blocks spaced 25-50m apart. Trap stations consisted of a Victor snap trap (baited with peanut butter) and a chew block placed 1-2m apart. Every other trap station was housed inside an unarmed Protecta bait station (Bell Laboratories, Madison, WI) for protection against adverse weather. Transects were checked for activity and rebaited every 3-4 days for 18-22 days. The total number of trap nights was calculated as one trap set for one night; traps sprung without capture were assigned a value of 0.5 trap night. Results from these surveys were compared with similar surveys conducted before bait application in August 2007 when eight trap station transects were placed in coastal and upland habitats (Buckelew *et al.* 2007a).

Rats in the Aleutians are more frequently detected in late summer (August-September) when densities are highest following the completion of peak breeding (Dunlevy and Scharf 2006); however, logistical constraints prevented monitoring during this period. Therefore eleven transects, totalling 335 chew devices, were placed on beaches before leaving the island in late June for prolonged detection through late summer. During 2010 all chew devices will be inspected for incisor marks and reset.

To evaluate the persistence of bait during winter, 3m radius circular plots ($28.3m^2$) were sampled for pellets nine months after bait application. Randomly located plots were sampled staking one end of a 3m string to the plot centre, and the observer counting pellets while walking in a circle at the distal end of the string. The mean number of pellets encountered in plots ($n = 466$) was used to extrapolate the amount of bait (in kg/ha) remaining in different habitats.

Mortality of non-target species

During May-June and August 2009, formal and informal carcass surveys were conducted for birds that might have died from exposure to rodenticide. Formal surveys involved searching for carcasses on 67 beaches (or beach segments) either once or multiple (3-5) times. Informal surveys were opportunistic encounters of carcasses made while transiting the island. All carcasses encountered were collected and stored at ambient temperature until later transferred to a freezer (1-8 days after collection). Testable tissues or carcasses were analysed for brodifacoum residues. Carcasses too old to have any testable organ tissue were removed from the island to reduce secondary exposure by avian scavengers.

We did not study natural "background" mortality at Rat Island prior to the bait application. Therefore, standardised beach carcasses surveys (Coastal Observation and Seabird Survey Team, (COASST)) data available from islands east (Adak Island, 350km distance) and west (Buldir Island, 175km distance) of Rat Island were used as a control to provide reasonable approximation of the diversity of birds expected by natural mortality.

RESULTS

Biological surveys

Mean detections per point count were similar in 2009 to prior years for Lapland longspurs (*Calcarius lapponicus*) and gray-crowned rosy finches (*Leucosticte tephrocotis*), but the counts for winter wrens (*Troglodytes troglodytes*)

Table 1 A comparison of abundance of bird species (mean ± s.d.) detected using standardised surveys conducted before and nine-months after (2009) the application of cereal baits containing brodifacoum (25ppm) on Rat Island.

Species	Pre	Post	t	P
Gray-crowned rosy finch[1]	1.3 ± 0.8	0.8 ± 0.8	1.993	ns
Gray-crowned rosy finch[2]	0.0 ± 0.2	0.0 ± 0.1	0.327	ns
Lapland longspur[1]	2.1 ± 0.6	2.6 ± 1.1	-1.963	ns
Lapland longspur[2]	4.5 ± 3.0	5.1 ± 1.9	1.444	ns
Winter wren[1]	2.6 ± 0.7	3.3 ± 1.0	-2.469	0.026
Winter wren[2]	0.1 ± 0.4	0.3 ± 0.7	2.197	0.030
Rock ptarmigan[3]	1.2 ± 1.4	3.6 ± 1.6	-7.186	<0.001
Glaucous-winged gull[4]	615	1027	-	-
Black oystercatcher[1]	0.2 ± 0.2	0.1 ± 0.2	-2.076	ns
Rock sandpiper[1]	0.5 ± 0.9	0.6 ± 1.1	0.825	ns

[1]Species detected using line transects (pre (2007-08), n = 32 with 5 repetitions, and post (2009), n= 16 with 5 repetitions).
[2]Species detected using point count surveys (pre (2007-08), n = 74; post(2009), n = 57).
[3]Species detected using swath transects (pre (2007-08), n= 56; post(2009), n = 52).
[4]Species detected using nearshore boat surveys (pre (2008), n =1; post (2009), n=1).

were slightly higher in 2009 than previously (Table 1). Similarly, counts on fixed beach transects showed no differences between 2009 and earlier years for longspurs and rosy finches, but were higher for winter wrens (Table 1). Rock ptarmigan (*Lagopus muta*) counts were higher in 2009 than in earlier years (Table 1).

Black oystercatchers (*Haematopus palliates*) and rock sandpipers (*Calidris ptilocnemis*) were detected in equal numbers to previous years (Table 1). While transiting the island, incidental observations were made of rock sandpiper and black oystercatcher nests. Seven black oystercatcher nests and six rock sandpiper nests were encountered.

Given differences in sampling methods, statistical comparisons of glaucous-winged gull (*Larus glaucescens*) estimates were not possible (Table 1). However, detection rates for all common bird species monitored in this survey, including glaucous-winged gulls, were as high or higher than prior to rat eradication.

Eradication confirmation monitoring

In August 2007, before eradication, there was a 38% trap success (TN = 362) and rat chews were detected on 39% of the blocks. Rat activity was highest in coastal compared to stream or inland habitats. Trap success was 59% in coastal areas (TN = 212) and chews were detected on 64% of the blocks. In June 2009, after bait application, no rats or rat sign were detected on trap station transects after 9068 trap nights. Similarly, no signs of rat activity were detected in 1550 nights of chew block detection effort.

All bait pellets in coastal habitats were either directly consumed or degraded during the winter (Table 2). Very few, highly-degraded pellets remained in upland and lake habitats, with an average of < 1 pellet per 100m^2. Pellets were laterally compressed and in an advanced state of decay, having lost their shape and integrity, after prolonged exposure to weather and snow.

Mortality of non-target species

A total of 422 bird carcasses were found during formal and informal surveys in May-June and August 2009 (Table 3). Of these, it is likely that some of the carcasses encountered during August were first discovered but not removed in June. Ninety one of the carcasses were submitted for analysis, and the results will later be made available for publication. The majority of carcasses collected were in moderate to advanced stages of decomposition.

Most carcasses were of glaucous-winged gulls, but a few carcasses of other species, normally encountered on beached bird surveys in the Aleutian Islands, were also found and are unlikely to be casualties of the operation (Table 4). A small proportion of the gull carcasses were scavenged, presumably by avian predators, and only the skeletons remained. Most of the bald eagle carcasses were around the coastal periphery of the island, either along the beach berm or near coastal streams. Eagle carcasses on the interior of the island were found close to lakes or streams, with the exception of a few that were in upland areas. No lethargic birds or birds exhibiting abnormal behaviour suggestive of exposure to rodenticide were observed during our surveys.

DISCUSSION

Biological surveys

Surveys conducted after eradication showed no evidence of a significant difference in detection rates for Lapland longspur or gray-crowned rosy finches, although high variability and low sample sizes made detecting any pattern difficult. There was evidence of a significant increase in the counts for winter wrens following the eradication of rats. The number of ptarmigan detected on line transects were 105% higher after eradication than before, indicating that there were no adverse effects of the eradication on ptarmigan abundance.

Table 2 Number of pellets detected (per plot and per ha) during May-June 2009 and the nominal number of pellets applied per habitat type during September-October 2008 on Rat Island.

Habitat	No of plots	No. pellets/ha applied (2008)	No. pellets/ha remaining (2009)	Mean pellets/plot	Kg/ha remaining	% diff
Coastal	88	*8180*	2	0.006 ± 0.054	0.005 ± 0.042	*99.9*
Upland	296	*4090*	43	0.122 ± 4.361	0.095 ± 0.409	*98.9*
Lake	82	*8180*	195	0.549 ± 1.982	0.427 ± 1.543	*97.6*
Total	466	*4090*	62	0.176 ± 0.978	0.137 ± 0.738	*98.5*

Table 3 The bird species and maximum number of individuals found dead on Rat Island during summer 2009 following the application of cereal bait containing brodifacoum (25ppm). Birds were encountered either opportunistically or during beach carcass surveys. Bald eagles and glaucous-winged gulls are listed in parentheses by age class (adult: subadult: unknown) (P= present, encountered but not enumerated).

Species	May/ June	Early Aug	Late Aug
Predominantly terrestrial species			
Bald eagle	43 (14:29:0)	2 (0:1:1)	1
Common raven	2		
Emperor goose	1		
Gray-crowned rosy finch	2	1	
Green winged teal	1		
Rock ptarmigan		1	1
Lapland longspur	2		
Peregrine falcon	1		
Snow bunting	2		
Predominantly marine species			
Common eider	2		
Glaucous-winged gull	222 (58:188:0)	57 (10:59:1)	41
Black-legged kittiwake		3	
Unidentified shearwater		1	
Harlequin duck	2	2	
Green-winged teal		1	
Northern fulmar	2	6	1
Parakeet auklet	1		
Pelagic cormorant or unk	2	1	1
Pigeon guillemot	2		
Tufted puffin	3		
Unidentified puffin		1	1
Thick-billed murre		1	1
Common murre		1	
Unidentified murre		1	
Unidentified auklet	1		P
Least auklet		2	
Whiskered auklet	1	1	

Table 4 A) The total numbers of bird carcasses found during COASST beach surveys conducted on Adak, Rat, and Buldir Islands during summer 2006-09. B) The total number of bird carcasses by species found during beach surveys conducted during summer 2009 (data source: COASST, accessed on December 2, 2009, http://depts.washington.edu/coasst/patterns.html). Total beach area surveyed on transects on Adak I. = 2.3 km, Buldir I. = 4.7 km, and Rat I. = 37 km. Numbers in ( ) refer to number of carcasses found per km of beach surveyed.

	Adak I. no seabird colony; rat-infested	Rat I. no seabird colony; rat- free	Buldir I. seabird colony; rat-free
A) Total bird carcasses found			
2006	0	na	32
2007	0	na	61
2008	Na	na	83
2009	0	235	57
B) Carcasses by species found during 2009 surveys			
Ancient murrelet	0	0	2 (0.43)
Black-legged kittiwake	0	0	7 (1.49)
Common eider	0	2 (0.05)	0
Common raven	0	2 (0.05)	0
Emperor goose	0	1 (0.03)	0
Glaucous-winged gull	0	214 (5.78)	13 (2.77)
Green- winged teal	0	1 (0.03)	0
Harlequin duck	0	2 (0.05)	0
Horned puffin	0	0	2 (0.42)
Laysan albatross	0	0	1 (0.21)
Northern fulmar	0	2 (0.05)	0
Parasitic jaeger	0	0	1 (0.21)
Parakeet auklet	0	1 (0.03)	1 (0.21)
Pelagic cormorant	0	2 (0.05)	5 (1.06)
Peregrine falcon	0	1 (0.03)	0
Short-tailed shearwater	0	0	1 (0.21)
Thick-billed murre	0	0	16 (3.40)
Tufted puffin	0	3 (0.08)	2 (0.43)
Unknown	0	1 (0.03)	6 (1.28)
Whiskered auklet	0	1 (0.03)	0
Total	0	235 (6.35)	57 (12.13)

It is not known how productive Rat Island was for nesting shorebirds prior to rat introduction, but the coastal and upland areas were not highly productive breeding habitat in 2007-2009. Nevertheless, while breeding was recorded previously, ours is the first record of chicks hatching for black oystercatchers and rock sandpipers. No specific surveys for burrowing nesting seabirds on rock stacks had been conducted in previous years; therefore, it is not clear whether four pigeon guillemots nests discovered on rock stacks were a response to rat removal. The location of at least one of the guillemot nests in the entrance of a burrow believed to have belonged to a rat (identified by small pile of chewed invertebrate shells located beneath a low rock overhang; a potential rat feeding station) is suggestive of recolonisation.

Eradication confirmation monitoring

Nine months after bait application, no rats were observed or detected and no bait remained on the coast. A few baits did, however, persist into the spring following their application in inland habitats. This observation of varying decomposition rates according to habitat is consistent with degradation trials conducted in the Bay of Islands on Adak Island, where pellets placed in inland habitats persisted longer than those in low altitude coastal habitats (Buckelew *et al.* 2007b). Pellets remaining on Rat Island were in the final stage of decomposition, and likely persisted due to the overwinter freezing conditions. After a summer of relatively warm temperatures and heavy rains it is likely that the few remaining pellets will dissolve entirely.

Mortality of non-target species

Mortality of individuals of some non-target species is often an unavoidable consequence of successful eradications. Some winter die-off of birds is not unusual in the Aleutian Islands, but the numbers of glaucous-winged gull and bald eagle carcasses observed during our surveys were substantially higher than expected. The numbers of other bird carcasses encountered, particularly for seabird species, were consistent with coastal observations on Buldir Island, the closest seabird colony for which data exist. Buldir Island is rat-free and contains large seabird breeding colonies (unlike Rat Island); therefore, it is not an ideal reference site. Nevertheless, the list of carcass species from Buldir provides a reasonable approximation of the diversity of birds that could be expected naturally on Rat Island, suggesting that many of the species found were unlikely to have been a result of the eradication campaign.

Gull and eagle carcasses tested positive for brodifacoum suggesting that most of these mortalities were due to primary and/or secondary exposure to rat baits. At the time of writing the toxicology results were unavailable for publication, and will be made available in a later report. The highly degraded state of most carcasses encountered suggests that the birds died many months before they were collected, probably soon after the baits were spread. Considering their advanced state of decay, most carcasses recovered during repeated visits to Rat Island during August 2009 were from beaches not surveyed during May-June or were carcasses encountered but not previously collected. Thus it is likely that some carcasses were double-counted during August, thus may have been an overestimate of mortality.

Secondary poisoning of bald eagles is presumed to have been from scavenging sick or dead gulls and rats. Gull feathers and rat remains (fur and bones) were found in several eagle boluses (Buckelew *et al.* 2009). This pathway of secondary brodifacoum exposure was not previously identified as a significant risk to eagles. Typically, eagles are absent from Rat Island during fall when they congregate around streams on nearby islands to feed on spawning salmon (Gibson and Byrd 2007). Eagles are most abundant on Rat Island during summer. We only recorded six eagles during the baiting operation suggesting that most eagles had already departed. Nevertheless, eagles apparently arrived on Rat Island later in the season (fall or early winter) and scavenged or preyed on gulls exposed to brodifacoum.

CONCLUSIONS

Eradication of invasive species has direct benefits to species impacted by non-native predators and indirect benefits to native ecosystems. However, there may be short term impacts on native species from the rodenticide, as observed on Rat Island. Eagles and gulls suffered unintended and unexpectedly high mortality which has resulted in a decline in the eagle population. The recovery of a native ecosystem on Rat Island is almost certain to provide prey resources sufficient for eagles to completely recover to former, or possibly higher, breeding densities. Methods to estimate gull populations on Rat Island were not consistent with those used after bait application, so we were unable to detect small changes with confidence. Available counts, however, did not suggest a population-level decline for gulls.

The removal of Norway rats and Arctic foxes (completed in 1984) should enable the recovery of communities on Rat Island similar to those present before the introduction of these non-native predators. Our data indicate that following rat removal recovery is beginning for species such as winter wrens. Additionally, we documented successful nesting by pigeon guillemots, rock sandpipers, and black oystercatchers. If Rat Island is now rat-free, 2009 was the last season in which species on the island were affected by rats. Continued monitoring in future years will further document ecosystem changes on Rat Island. We anticipate that there will be increased densities of land birds that were previously preyed on by rats, recolonisation of the island by burrow-nesting seabirds, and changes in the vegetative and intertidal communities.

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Eradication of feral goats (*Capra hircus*) from Makua Military Reservation, Oahu, Hawaii

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Abstract Feral goats (*Capra hircus*) were a significant threat to the native habitat and endangered biota unique to the Makua Military Reservation (MMR) on the island of Oahu, Hawaii. The Oahu Army Natural Resource Programme (OANRP) was tasked with the removal of these animals. From December 1995 through February 1997, ground hunts were undertaken by contract hunters from the U. S. Department of Agriculture Wildlife Services while plans for a fence to enclose the 1695 ha MMR were finalised. In 1996-1997, the first stretch of fencing separating MMR from a public hunting area was completed along with fencing around the eastern perimeter of the valley. Contract and staff hunts continued along with snares until the last portion of the fence was finished in 2000. OANRP staff then employed other techniques to complete the eradication, including Judas goats and aerial hunting. When the last goat was eradicated in July 2004, a total of 1565 goats had been destroyed using a combination of techniques.

Keywords: Military training areas, fencing, Judas goats, snares, aerial hunting

INTRODUCTION

Threats to the integrity of native ecosystems from feral ungulates, such as goats (*Capra hircus*) and pigs (*Sus scrofa*), have long been recognised in Hawaii and other islands (Spatz and Mueller-Dumbois 1975; Vitousek 1988; Atkinson 1989; Cuddihy and Stone 1990; Desender *et al*. 1999). These animals can radically alter entire native habitats, as well as jeopardising the component species. They browse on almost any type of vegetation, including native grasses, shrubs and small trees, as well as the seedlings of any life form, which can lead to overgrazing and result in primary and secondary impacts to ecosystems (Campbell and Donlan 2005). These impacts lead to the loss of native biodiversity, the degradation of native ecosystems, acceleration of soil erosion and the colonisation by herbivore resistant non-indigenous weeds. Ground-level ferns, herbs, saplings and shrubs are the plants most susceptible to ungulate damage (Sakai *et al*. 2002). Goats have a very efficient digestive system, a low metabolic rate, and can tolerate very arid environments, which allows them to thrive in habitats unsuitable for many other animals (Silanikove 2000). Goats can be found in extremely steep, rugged terrain, a matter of particular concern because many rare and endangered plants are now restricted to these otherwise inaccessible areas. The native flora and fauna of Hawaii evolved in the absence of large herbivorous mammals. As a consequence, the endemic flora appears to have lost natural defences against herbivory (e.g., Vitousek 1988; Atkinson 1989; Primack 1993; Paulay 1994). Results from Bowen and Van Vuren (1997) support this hypothesis and corroborate the belief that human introduced herbivores are a major contributor to island extinctions. Thus feral ungulate management is one of the primary priorities for any restoration project in Hawaii.

The O'ahu Army Natural Resource Programme (OANRP) is responsible for managing 50 species of endangered plants, eight of species endangered animals, and the ecosystems upon which they depend in U. S. Army training areas on O`ahu. The legal requirement driving the Army's ecosystem management programme is the Endangered Species Act (ESA) Sections 7(a)(1) and 7(a)(2). These sections of the ESA require that Federal agencies use their authority to conserve federally listed species, and ensure that their activities are not likely to jeopardise the continued existence of any federally listed species.

This paper documents how we conducted an eradication programme in a "mainland island" formed by the U. S. Army's Makua Military Reservation (MMR) on the island of O'ahu in Hawaii, USA.

STUDY AREA

MMR is 1695 ha and is the US Army's largest manoeuvre/live-fire training area on O'ahu, Hawaii (Fig. 1). It encompasses two gulches, Kahanahāiki and Mākua, which are the northernmost major valleys on the leeward side of the Wai'anae Mountains (Fig. 2). The terrain at MMR varies from a gradual to moderate valley bottom and sides that increase in steepness with elevation, becoming extremely steep, exposed, and rocky above about 360 m. Elevations range from sea level to approximately 1000 m. While most of the natural habitats within MMR are highly disturbed with large expanses of alien grassland in the lowlands, there are large pockets of primarily native dry

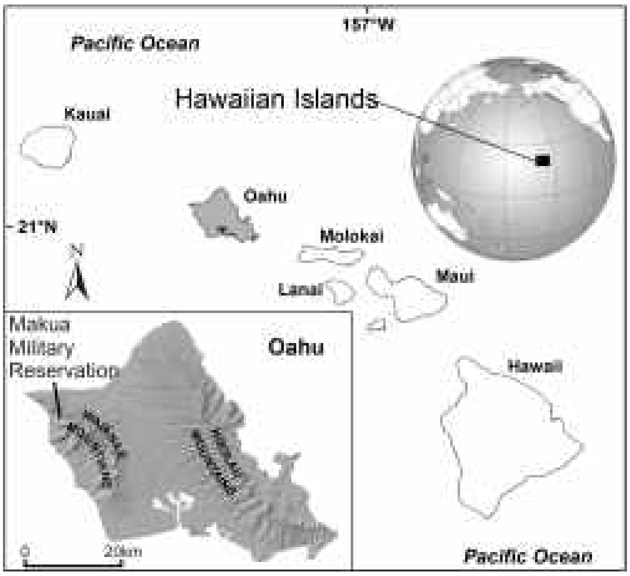

Fig. 1 The Hawaiian Islands and the Makua Military Reservation on O'ahu.

Pages 280-284 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

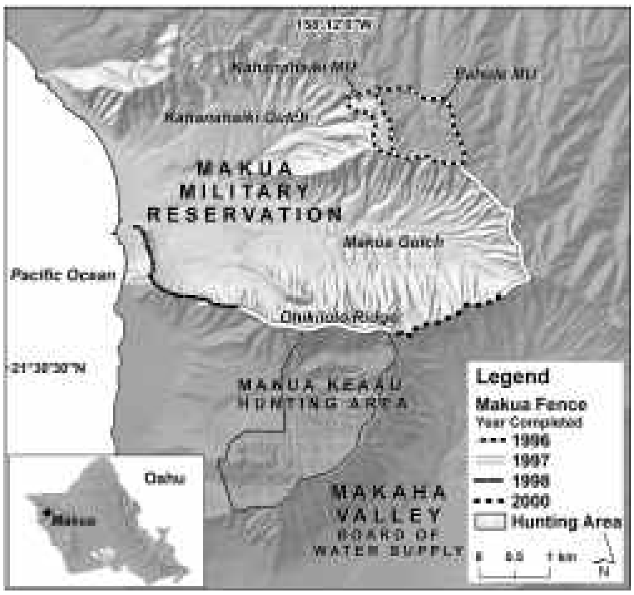

Fig. 2 Fences constructed by year at Makua Military Reservation (MMR).

and mesic forest dominated by *Diospyros sandwicensis*, *Diospyros hillebrandii*, and *Metrosideros polymorpha*. There are also large expanses of native dry cliff vegetation, ridge-tops with mesic native shrub land and forest, including areas dominated by *Dodonaea viscosa* and *Metrosideros tremuloides*. There is one rare natural community, the *Pritchardia kaalae* lowland mesic forest.

The Mākua Kea'au public hunting area, Mākaha Valley and 'Ōhikilolo ranch are adjacent to the southern border of MMR (Fig. 2). These areas contain large numbers of goats as there is little population control. Without a barrier to prevent ingress, feral goats would migrate over the long southern ridge of MMR ('Ōhikilolo). Due to military training and unexploded ordnance (UXO) public hunting is not allowed in MMR. Furthermore, other access to the area is restricted to times when there are no military activities.

METHODS

In order to eradicate all of the feral goats from MMR, we employed a multi-faceted approach throughout the campaign (Fig. 3). To eliminate ingress from the high density goat population to the south, a fence was constructed

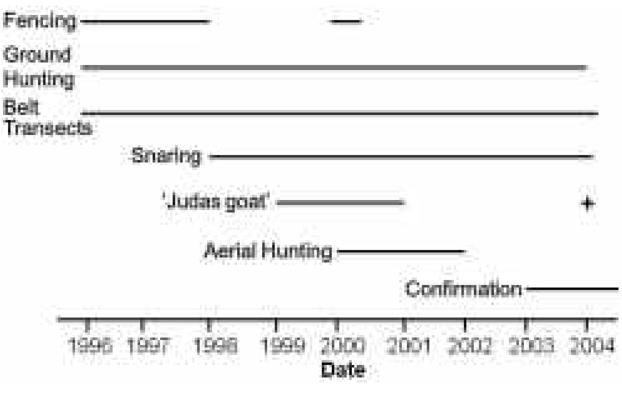

Fig. 3 Timelines of methods employed during goat eradication operations at Makua Military Reservation. The star denotes the time of the last "Judas goat" deployment.

in five phases. The fencing was coupled with ground hunting, using a combination of contractors and staff. Three 500 m ungulate-sign, belt transects were installed to detect tracks and/or scat (goat sign) to monitor the success of the eradication effort. As fence construction came to an end and goat numbers decreased, three other control techniques were employed to increase the removal rates: snares, aerial hunting, and ground hunting using radio-collared Judas goats. The final phase of the eradication was confirmation of the absence of goats.

Fence construction

Fencing materials used were: 1) 87 or 122 cm tall, graduated mesh pattern, galvanised, hinge lock woven wire fence; and 2) either an 87 or 132 cm × 4.88 m, 4 gauge, graduated mesh pattern, galvanised fence panels. Terrain dictated which type of fencing was used.

Ground hunting

Ground hunting with staff and contractors from the U.S. Department of Agriculture Wildlife Services (WS) began in December 1995 and continued through to July 2004. Hunting teams consisted of 2-4 groups of 2-3 people. Teams split up onto separate consecutive ridges spotting for each other. A variety of calibres (.308, .270, .223) and actions (bolt, lever, semi-auto) of firearms were used depending on the preference of the hunter. Ammunition ranged from 150-180 grain. All personnel wore blaze orange so they were visible from a distance and carried two-way VHF FM radios in order to communicate with each other and with the Army's Range Control at MMR.

Snaring

In 1998, customised multi-strand, aircraft quality steel cable snares were obtained from the Raymond Thompson Snare Co. (Lynwood, WA). They were placed along narrow trails with the noose suspended at 75-125 cm from the ground. The size of the suspended nooses ranged from 25-40 cm diam. In order to asphyxiate the animals quickly, all snares were placed in steep areas so that footing would be lost and unable to be regained.

Aerial hunting

Aerial shooting operations were conducted from 2000-2002 using a Hughes 500D helicopter with one shooter aided by spotters on the ground. Pilots and shooters were experienced and certified by the U.S. Department of Agriculture for this type of operation. The shooter used a Benelli semi-automatic 12 gauge shotgun with 00 buck shot.

Judas goats

In 1999, we attempted to use "Judas goats" (Taylor and Katahira 1988) to track goat movements and locations and determine herd associations in MMR. Four goats were fitted with Telonics (Mesa, AZ) VHF MOD500 transmitters that emitted a unique radio signal. Transmitters could be tracked from the ground or air using a Telonics TR2 telemetry receiver with a Telonics RA-2AK (Yagi-Uda) "H-Type" 2-element antenna. The first two goats released were domestic animals purchased from a local ranch (1 female and 1 immature male) and with a white coat to facilitate later sightings. The other two goats (immature males) were live captured in MMR using modified snares.

In 2004, we contracted WS to capture goats in Kea'au using a net-gun from a helicopter. Two animals were captured; one was fitted with a Telonics VHF MOD500 transmitter and the other with a satellite GPS receiver. Both goats were then released.

Transects

We used three belt transects to monitor changes in feral goat sign over time. Transects were 500 m long × 5 m wide. Monitoring stations were tagged and labelled every 10 m along each transect. Observers recorded all ungulate sign, including feeding, scat, and trails for goats within each of the 10 × 5 m transect sections. Only presence/absence data was taken and no measures of the overall density were measured within the plots.

RESULTS

Fence construction

Fence construction at MMR began in 1996 with the work done by the Hawaii Natural Area Reserves System staff, remote fencing service providers from Hawaii Volcanoes National Park, and John Hinton and Southwest Fence and Supply Co. Inc. The fence followed the upper reaches of Kahanahāiki and Pahole gulches, which enclosed a 2 km portion of the northeast rim. In 1997, the fence was extended along the northeast rim and about 500m down ʻŌhikilolo. This was built in conjunction with the initial 2 km of fence on ʻŌhikilolo, which headed seaward from the highest point. In 1998, the seaward section of the fence on ʻŌhikilolo was completed. The fencing material for all of these sections was 122 cm tall, graduated mesh pattern, galvanised, hinge lock woven wire fence. In 2000, the final and most treacherous portion of the fence was completed to close the gap along ʻŌhikilolo ridge. We used 132 × 490 cm, 4 gauge, galvanised fence panels for this section because of the rugged terrain. These rigid panels are portable and can be cut and manipulated to fit the landscape. In total, 12 km of fencing was erected around MMR. This completely isolated the goat population in MMR from the adjacent populations to the south but did not encompass the entire valley as there are no populations of goats to the north (Fig. 2).

Ground hunting

When military training commenced, access for hunting was forbidden. In 1997, MMR was used quite extensively by the military for training purposes. A series of range fires closed MMR to training from 1998-present, which enabled the eradication campaign to be completed. Some areas were also of limited access or off-limits due to UXO. All ground hunts were escorted by an UXO technician to identify potential hazards. Staff were also required to wear Kevlar flak jackets and helmets as a precaution.

A total of 560 hunter days (4478 hunter hours) were required for 1232 goats removed by hunters. For simplicity, the very small number of animals and hours from December 1995 were combined with the total for 1996. From 1996-1999, ground hunting removed a large percentage of the animals in MMR (Fig. 4). An average of 2.2 staff hours/goat removed was observed during this period. From 2000-2004, more time was spent searching and the effort required per kill increased twenty-fold to an average of 44.8 staff hours/goat removed.

Fig. 5 Total number of goats removed (bars) and snaring effort of staff (line) by year during the MMR eradication campaign. The numbers above each bar represent the average number of staff hours expended per goat each year.

Snaring

Snares were set in 17 clusters of 20-40 snares apiece throughout the head of the valley. After the initial set, snares were checked on subsequent trips for catches and condition, then reset or removed as needed. New snare clusters were installed when animals were seen moving into new areas. In total, 336 snares were set logging about 559,440 snare hours. The total effort required 1064 staff hours and removed 163 goats (Fig. 5). From 1998-1999, snaring required an average of 4.2 staff hours/goat removed. As goat numbers decreased, more effort was required to increase the number and location of snares so the mean increased to 18.1 staff hours/goat removed.

Fig. 4 Total number of goats removed (bars) and ground hunting effort (line) by year during the MMR eradication campaign. The numbers above each bar represent the average number of staff hours expended per goat each year.

Fig. 6 Total number of goats removed (bars) and aerial hunting effort (line) by year during the MMR eradication campaign. The numbers above each bar represent the average number of staff hours expended per goat removed each year.

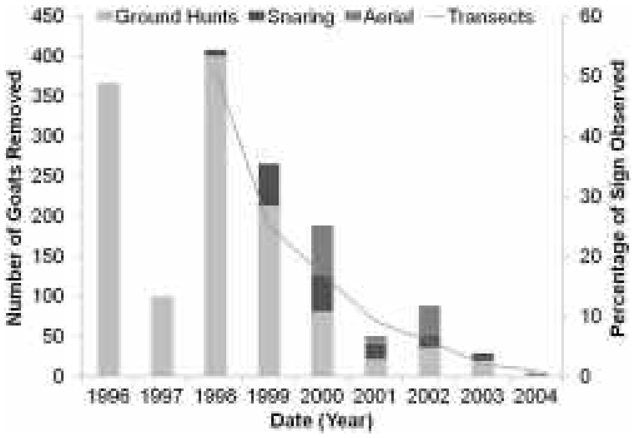

Fig. 7 Total number of goats removed with all removal methods used combined by year. The line represents the progression of the goat eradication over time, which was a measure of the percentage of sign observed along transects.

Aerial hunting

The effort required for this part of the operation was 2.0 staff hours/goat in 2000 (Fig. 6) because up to nine spotters were used each time, many of which were flown into position. As goats became shy of the helicopters, the effort required increased to 9.3 staff hours/goat in 2001, even though we decreased the number of spotters in the field. In 2002, we expanded the aerial hunts to the north side of Mākua Gulch after goat herds were observed there and further decreased the amount of spotters in the field. This decreased the amount of effort required to 1.7 staff hours/goat removed. The mean effort required for all three years was 4.4 staff hours/goat removed. We combined UXO technician escort, shooter, and spotter hours for this total. Overall progress of the eradication campaign was indicated from sign along the transects (Fig. 7).

Judas goats

The 1999 deployment of Judas goats was unsuccessful. The two white domesticated goats did not move from their drop point for almost two years until one jumped over the fence into Kea'au and the other herded up with a nanny and kid. These three were subsequently shot. However, the wild-caught Judas goats immediately united with others and we were able to track them down to eliminate their associates. After this, we found it very difficult to locate either animal easily as they strayed from the original snare spot. We were able to approximate their location but due to difficult terrain and access, visual verification was impractical.

The 2004 deployment was unsuccessful as well. The radio collared individual was able to escape back over the fence into Kea'au and the satellite collared one was snared soon after release. It was not unexpected for animals to leave MMR as the fence was constructed with high spots on the inside to allow escapes. High points were strictly avoided on the outside making the fence permeable in one direction.

DISCUSSION

In any eradication campaign, immigration must be eliminated. In our case, ~8 km of fencing was needed to create a "mainland island". The fence took four years to complete with the last section in very rugged country where safety lines and rappelling were necessary during construction. Once immigration by goats ceased, most of the animals were eliminated before the fence was completed. Constant upkeep of the fence is necessary, so we conduct quarterly inspections. The environment in MMR is very harsh with constant salt spray, high/gusty winds with a dusty/gritty substrate, solar radiation and occasional fires. All of these environmental factors have taken their toll on the integrity of the fence, especially the seaward sections.

In 1998, we experimented with snares as control option in conjunction with ground hunting. Although they are controversial because of concerns over animal welfare, snares are cost effective and efficient for feral pig control (Anderson and Stone 1993; Hess *et al*. 2006). They are small, light weight, and simple to erect, making it easy to set out a large number in a short period of time over multiple areas. Unlike any of the other management tools used on this campaign, snares work 24 hr/day seven days/week. The designation of MMR as off limits for hunting allowed for the extensive use of snares, which effectively removed goats after their populations were reduced by ground hunting. The first snares were installed in December 1998 and numbers were increased in 1999, when ground hunts were still quite effective. The percentage of goats snared was only 2% in 1998 and 20% in 1999. By 2000, ground hunts were becoming less effective so the percentage of goats snared gradually increased from 26% in 2000 to 75% in 2004. The mean percentage of goats removed from 2000-2004 was 43% for both ground hunting and snaring but the effort (staff hours/goat) was over half for snaring (18.2/44.8).

Aerial hunting was also effective method of removal, particularly since it allowed shooters access to goats in areas that were inaccessible to the ground based hunting and snaring. The helicopter was also able to cover the entire range in a couple of hours. The mean percentage for animals removed via aerial hunting was 30% from 2000-2002, while the mean effort required was only 4.4 staff hours/goat. This method was quite effective when compared to ground hunting (42% at 14.9 staff hours/goat) and snaring (29% at 14.9 staff hours/goat) during this same time frame.

In contrast, ground based radio-tracking of "Judas goats" (Taylor and Katahira 1988; Rainbolt and Coblentz 1999; Campbell 2002) in MMR was problematic. There appeared to be association issues between goats that were purchased or captured offsite and the goats already present. These same association issues have been observed in other eradication campaigns such as Sarigan Island in the Northern Mariana Islands; Desecheo Island, Puerto Rico; Tasmania; and West coast of south island, NZ (Howell and Atkinson 1994; Kessler 2002; Karl Campbell pers. comm.). The steepness and rocky terrain appeared to cause the radio signal to create an echo, simulating a false location. The simultaneous use of snares had a direct impact on the survival of at least one collared goat. WS shooters or trackers were unable to utilise the "Judas goats" in any of their aerial or ground based operations to verify these issues. It would have been preferable to test this method from the air to see if the applicability would have been worth the cost.

Prior to the completion of the seaward section of fence in 1998, an unsuccessful goat drive was attempted using a helicopter piloted by an experienced pilot/rancher. The Wai'anae community expressed their concerns about the eradication techniques and wanted to explore another "non-lethal" option. No animals were removed using this technique but it likely educated goats to the helicopter as a threat.

We found that flexibility of multiple eradication methods was a key to the eradication of goats from MMR. As the effectiveness of one method diminished other methods were employed in order to prevent the population from learning to avoid specific techniques. When multiple management methods were combined, goat removal rates were higher than if only one method was employed. Selecting the timing of the eradication methods employed is always challenging. Other successful eradication campaigns found that ground hunting followed by aerial hunting was successful (Rainbolt and Coblentz 1999; Kessler 2002; Campbell *et al*. 2004; Campbell and Donlan 2005; Cruz *et al*. 2009). In our campaign, this same progression of methods worked well. The addition of snaring increased the effectiveness of the eradication campaign at a crucial time when goat numbers were low and "Judas goats" were found to be ineffective. Without the use of snares, it is likely that the eradication campaign would have required a longer period of time.

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Eradication of alien invasive species: surprise effects and conservation successes

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Abstract The growing number of biological invasions worldwide is now being accompanied by burgeoning successful alien species eradications on islands of increasing size, topography and habitat complexity. However, the extent of these achievements depends on the definition of success. In most cases, success or failure are measured in terms of the absence or presence of the target alien species. It is becoming increasingly evident that how the invaded ecosystems respond to eradications should also be assessed. This is because some eradications have been accompanied by unexpected population explosions of hitherto seemingly harmless (or undetected) introduced species, previously suppressed by the eradicated alien species. These unexpected chain reactions are sometimes referred to as "surprise effects". We conducted an eight year study of plant and animal communities in a simple insular ecosystem invaded by ship rats (*Rattus rattus*) and domestic mice (*Mus musculus*). We assessed these communities for potential surprise effects following rodent eradication. Next we eradicated the rats and mice following a protocol tailored to the presence of other introduced species. We then continued to monitor changes to the ecosystem, a step too often missing after eradication programmes. We then assessed the success of our eradication in terms of: 1) absence of the eradicated species; 2) recovery of the ecosystem; and 3) absence of surprise effects.

Keywords: *Rattus rattus*, *Mus musculus*, *Achyranthes aspersa velutina*, trophic relationships, island conservation, control strategy

INTRODUCTION

Increased success with alien species eradication from islands is probably one of the major achievements of the last decade in conservation biology (Courchamp *et al*. 2003; Genovesi, 2005; Brooke *et al*. 2007; Genovesi and Carnevali 2011). An expanding number of species of plants and animal are now successfully - sometimes routinely - removed from islands that are increasingly large, rugged and complex. In particular, islands that only ten years ago were regarded as ineligible for alien invasive mammal eradication because of low feasibility are included in large-scale multispecies removal programmes (Courchamp *et al*. 2003).

Despite the increasing range of invasive species eradicated from islands, there has not been a parallel increased understanding of the ecological effects of such eradications. Instead, there is still a disconnection between these management programmes and studies of their consequences at the ecosystem level. Generally, removal of a pest species has undisputed benefits to the extant native biota, but empirical observation shows that these benefits can vary dramatically and unpredictably, and there may even be unexpected adverse consequences (Courchamp *et al*. 2003).

Exotic species interact with native species as well as among themselves, creating complex direct and indirect effects involving competition, predation and facilitation that can be difficult to comprehend, let alone to predict. For example, the removal of one exotic species can favour the expansion of others that were previously suppressed by the species removed. Thus, in addition to improving our abilities to eradicate exotic species, it is also important to characterise their role in invaded trophic webs in order to avoid these unexpected or "surprise effects". An illustration is the removal of herbivorous aliens such as rabbits and goats, which can lead to a release of exotic plants. In the absence of browsing, the exotic species may then out-compete native plants, leading to an explosion of weeds. In one such example on Sarigan Island (Mariana Islands), goats and pigs were removed in order to reverse the loss of forest, reduce erosion, and protect endangered native fauna (Kessler 2002). However, the removal of alien mammals allowed the introduced vine *Operculina ventricosa* to thrive and spread so rapidly, part of the island became overgrown by vines, with unknown consequences for the future of the whole ecosystem. Introduced mammals had previously held the vine at such low density that pre-operation monitoring did not reveal its presence. There are other examples with different trophic relationships (e.g., prey-predators or competitors, Courchamp *et al*. 1999; Caut *et al*. 2007). These surprise effects are not the rule, but as they may lead to additional ecological damage, it is important to anticipate them. The outcomes of change within these already perturbed trophic webs are not entirely intuitive and intervention as dramatic as species eradication should, where necessary, be preceded by careful empirical and theoretical studies of the whole ecosystem. Sometimes, the presence of a few individuals of a species that may appear of minor importance can mask powerful interspecific interactions.

Here, we describe a long-term project on Surprise Island (New Caledonia). Our goal was to define a rational methodology to manage invasive populations in insular ecosystems where there may be surprise effects when an introduced species is eliminated. Specifically, our approach followed three successive steps. First, we undertook complete floristic and faunistic surveys of the island. We also studied diet of the focal introduced species, which was the ship rat (*Rattus rattus*), a major invasive species, (Jones *et al*. 2008), that had allegedly been on Surprise Island for several decades. We also undertook demographic studies of key species in order mainly to assess population sizes of species most likely affected by the rats. This allowed us to develop hypotheses about trophic webs and the direct or indirect effects of the focal alien invasive species.

The second part of our programme was to construct and analyse mathematical models of the dynamics of populations that interact within the trophic webs reconstructed from our field studies based on parameters from data obtained in the field (see Courchamp and Caut 2005; Caut *et al*. 2007). These models presented a number of possible consequences of the elimination of the rats, focussing on representative tri-specific sub-systems, including potential surprise effects. Once we established the different system response possibilities, we eradicated the rats according to the methods and strategies dictated by the field conditions and predictions from the models (Caut *et al*. 2009).

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The third part of this study was long-term post-eradication monitoring of the entire ecosystem. In the present paper, we focus on steps one and three. We briefly outline our field methods and the insights these provided into changes of the ecosystem four years after rat eradication. We show how even the most careful programmes may struggle to avoid all repercussions of the removal of introduced species as pervasive as ship rats.

MATERIALS AND METHODS

Field site

The Entrecasteaux reef is approximately 230 km from the northern end of the main island of New Caledonia and constitutes four main islands, among which is Surprise (Fig. 1). This uninhabited island is ovoid, (about 800 m x 400 m), with a coast length of nearly 1800 m and an area of 24 ha. Each year, four years before the rat eradication (in 2005) and five years subsequently, we visited the island in November to assess the characteristics and short-term change of the plant and animal communities. Specifically, we collected data on: plant cover (different species), seabird abundance (different species), skink abundance, insect abundance (different families) and rat abundance/presence. We mapped the entire island, using a Thales GPS 6502sk/mk, focusing on the extent of the main vegetation units (about 25,000 GPS points). The GPS also provided geo-referenced points for year-by-year comparisons. Rat diet characterisation was performed with classic stomach content and faeces analyses as well as stable isotopic analyses. We will here provide information only on aspects directly relevant to plant communities. Additional details about the island and its ecosystems are provided elsewhere (Caut *et al*. 2008, 2009; Watari *et al*. 2011).

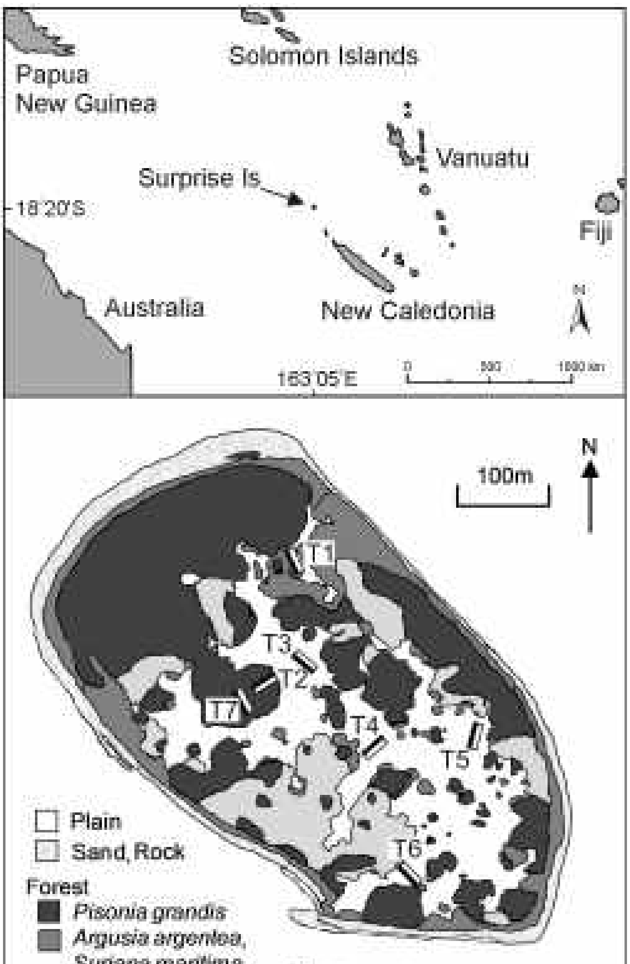

Fig. 1 Surprise Island showing the main vegetation units as well as the seven vegetation transects used to monitor the plant community changes. T1 to T7 are the seven plant transects. See key of the figure for more details.

Characterisation of the vegetation

We characterised the main vegetation units using: 1) five "plant plots" in each habitat unit within which species were identified in 20x20 m squares to assess the cover of each species present; 2) seven point-scale transects of 20 m to assess the cover of each species at different heights (Mueller-Dombois and Ellenberg 1974); and 3) geo-referenced annual photopoints for visual comparison of the plant communities. Samples of all plant species were collected for later identification of plant parts in rat stomach contents and faeces. In addition to constant visual observation, rats were regularly live-trapped along pre-established transects during yearly field sessions starting in 2001 and until their eradication in 2005. Details about the various vegetation types are available elsewhere (Caut *et al*. 2009).

Study of the rats' diet

Captured rats were killed, the stomach contents and faeces were removed and washed, and the fragmentary material obtained was compared with microphotographic reference collections of the epidermal tissues of Surprise Island plant species (120 different items) and animal prey. The relative contributions of plant items and animal prey were estimated for each stomach and faecal sample with a binocular microscope. Samples from livers of captured rats and samples from potential rat food items were collected for stable isotope analysis (Caut *et al*. 2008). Because the island was small and the vegetation types rather spread out and intermingled, we did not relate the diet to habitat. Too few individual mice were trapped for a quantitative diet analysis. Available data indicated, however, a potential overlap of diet, and a potential competition for watery plants (Caut *et al.* 2007).

Eradications

Given its size, eradication of rats from Surprise Island by trapping, as initially planned, would require 400 trapping stations on a 25 m grid (Pascal *et al*. 1996). However, we then discovered domestic mice (*Mus musculus*) on the island, which could undergo a population explosion should the rats be suddenly removed (Caut *et al*. 2007). This led to a changed rat eradication protocol to include the simultaneous removal of mice. Mice have been eradicated with bait stations at 25m on Mana Island (Hook and Todd 1992), but with their dominant competitors, ship rats, present on Surprise Island, mouse foraging ranges would likely be restricted. We calculated that eradication of mice by trapping would require a grid with trap and bait tubes every 5 metres; a total of 9800 stations over this small island. In addition to the cost and weighty logistics, this trap density would require significant damage to the plant communities and a major disturbance of seabirds. In addition, the numerous hermit crabs (*Coenobita* sp.) could lower trapping efficiency (or increase its cost), because the crabs can climb into bait stations to get the bait, and trigger traps. These logistic difficulties led us to switch from trapping to chemical control.

We used an anticoagulant poison that is target specific, will not affect other vertebrates, is harmless to invertebrates, and is widely used in France for rodent eradication. We used rodenticide bait blocks (3x3x1 cm, 25g) containing 0.005% bromadiolone (second generation anticoagulant toxicant), which is effective against rats and mice. Bait blocks were covered with paraffin wax to prolong their durability in a wet climate. We hand distributed the baits across the total surface of the island on a grid of 5x5 m. For access, we cut 38 transects (one every 15 metres) across the island (15 km of transects in the vegetation). On each of these 38 transects and every 5 metres, we dropped one bait block and tossed one at 5 metres to the left and another to the right. We repeated this process on days 0, 6, 11, and 18. About 950 kg (~40kg/ha) of rodenticide baits were used in total (250kg/session, ~11kg/ha). In parallel, traps were used to monitor rat activity just prior to, during, and after the eradication campaign (see also Caut *et al*. 2009).

Post-eradication surveys repeated the same methods used for all the ecosystem units (plants and animals) as in the pre-eradication phase (Caut *et al*. 2008, 2009).

RESULTS

Characterisation of vegetation

Our data revealed four contrasting vegetation units: 1) a ring of shrubs around the island dominated by 1 to 3 m high *Argusia argentea* and *Suriana maritima*; 2) a monospecific arboreal stratum of 3 to 10 m high *Pisonia grandis*; 3) scattered, dense patches of 1 to 3 m high *Scaevola sericea*; and 4) a central plain with more than a dozen main herbaceous species. Spatial coverage of the plant species present in each main vegetation unit based on plant plots and the point-scale transects is illustrated in Fig. 1. A limited stand of *Cassytha filiformis*, which is a potentially invasive plant native to Florida, was present on the island. Another notable exotic plant was *Colubrina asiatica* which was widely distributed over the island, although not dominating the vegetation cover.

Studies of the rats' diet

Rat digestive tracts contained 5202 identified fragments, 77% of which were of plant origin and included 17 of the 29 species of plants found on the island. *Pisonia grandis* was the most consumed plant (mostly as leaves), with 23% contribution of digestive tract contents and 74% presence in faeces of individuals (Fig. 2). Poaceae (grasses) contributed almost 11% to the diet of rats. About 18.6% of the stomach contents remained unidentified. Although widely distributed over the island, *Achyranthes aspersa* var. *velutina* amounted to only 4.67% of the rats' diet. We do not know how much this plant contributed to the diet of mice.

In total, animal remains formed 22% of the items present in the stomach contents (see also Caut *et al*. 2009). A significant component (35%) was ants, among which the only local species, *Pheidole oceanica*, was the most abundant. Ants found in rat stomach contents may have been ingested with the peanut butter bait, which attracted ants. If this were to be the case, ants would not have been a normal prey item of rats.

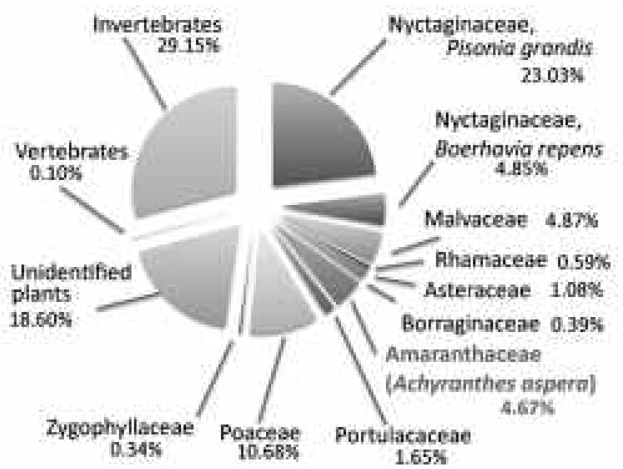

Fig. 2 Proportion of each item found in the stomach contents of rats invading Surprise Island. Note that *Achyranthes aspersa velutina* represents only 4.67% of all fragments found and are therefore not a major food item.

Eradications

After the eradication, trapping, tracking tunnels, wax tags, and hair tunnel devices deployed over the island confirmed the absence of rats on Surprise Island. Mice were eradicated at the same time as rats. If we follow the convention of confirmed absence for two consecutive years, we can claim a successful rodent eradication because both species have now been continuously absent for four years. Given the small size of the island and its remoteness, any rats or mice discovered in the future will most likely have come from a new introduction rather than from unnoticed survivors of the eradication programme.

The stand of *Cassytha filiformis* was removed to prevent post-eradication spread. Removal was not attempted for *Colubrina asiatica* due to its wide distribution over the island. Ant communities were left untouched as the local species predominated over the eight alien ant species in the two major habitats on the island (Cerdà *et al*. 2011): *Scaevola* shrubs and central plain. Furthermore, since *Pheidole oceanica* was the species most often eaten by rats, it was also the species most likely to increase in abundance. We did not witness any post-eradication spread of *Colubrina asiatica.* In contrast the indigenous *Achyranthes aspersa* became visibly more prominent over large parts of the island (Fig. 3).

Fig. 3 Georeferenced photos of the central plain of Surprise Island, in 2002 (left side, three years before the rat eradication) and in 2009 (right side, four years after the rat eradication). The dramatic growth of *Achyranthes aspersa velutina* is clearly visible.

Based on the yearly surveyed transects, simple statistical comparisons from 2002 (before rat eradication) and 2009 (after rat eradication) showed that *Achyranthes* covered more space (U = 3 ; p = 0.0060, Fig. 4a), was taller where it was present (U = 57 ; p < 0.001, Fig. 4b), and was more abundant than the other plants (Yates corrected Chi-square $\chi^2 = 826.18$ p < 0.001 df = 1, Fig. 4c) in the absence of rats compared to when rats were present.

DISCUSSION

Our long-term study of a small and remote island with a simple ecosystem enabled us to predict and avoid competitor release of domestic mice and a potential upsurge of the introduced *Cassytha filiformis.* We also found no evidence of an explosion of another introduced plant, *Colubrina asiatica*, or of the several species of exotic ants. It is possible that ant community structure has

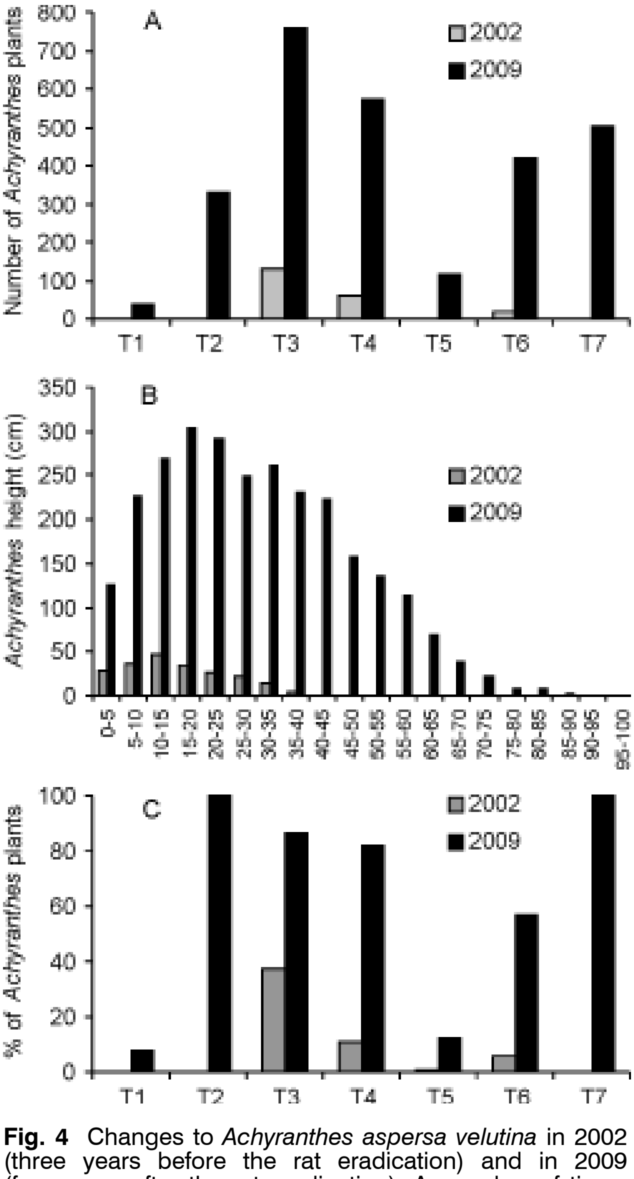

Fig. 4 Changes to *Achyranthes aspersa velutina* in 2002 (three years before the rat eradication) and in 2009 (four years after the rat eradication). A: number of times *Achyranthes aspersa velutina* was counted along the seven transects, showing that the plant was more abundant after rat eradication than it was before. B: height classes of *Achyranthes aspersa velutina* summed for all seven transects. This plant is on average taller after rat eradication than it was before. C: Proportion of *Achyranthes aspersa velutina* among all the plants present in the first metre of vegetation in the seven transects. This plant has outgrown the other plants since rat eradication.

Fig. 5 Brown booby (*Sula leucogaster*) in *Achyranthes aspersa velutina* on Surprise Island. Photo by Yuya Watari, Nov 2009.

changed, but no invasion has been observed. Following the rodent eradication, a local plant, *Achyranthes aspera velutina*, dramatically increased in height and coverage over the open spaces of the island and beneath *Pisonia grandis*. This was a very serious concern at first, as it was suspected that it could be the alien invasive *Achyranthes aspera* var. *aspersa*, released, directly or indirectly, by the rodent control. Positive identification of the plant as the local plant, which is heliophilous and generally the first to colonise after disturbances such as fire or cyclones (J.-Y. Meyer pers. comm.), suggests that the current explosion is normal and transitory. Seabirds may help disseminate the seeds of *Achyranthes aspersa velutina*, which stick to feathers (Fig. 5). Birds nesting on the ground in the central plain may in future be constrained by by this plant should its spread continue.

We hope that the increase now being observed is part of a normal phase of expansion following disturbance and that it will be followed by a return to previous conditions or something similar.

The basic requirements for restoring an invaded island are relatively well known (e.g., Parkes 1990; Veitch and Bell 1990; Towns and Ballantine 1993; Towns *et al*. 1997; Atkinson 2001; Saunders and Norton 2001; Courchamp *et al*. 2003; Brooke *et al*. 2007). In addition to these, pre-eradication studies and post-eradication monitoring are important components of success. Removing any species from an ecosystem can have diverse desired and undesired consequences, so it is crucial to quantify and predict these effects. Indeed, the quantification of desired effects can lead to improved control methods as well as a better justification of control programme for biodiversity conservation. Adequate knowledge can also help predict and thus prevent undesired or previously unexpected effects. We strongly believe that criteria for the success of invasive alien species eradications should include the subsequent recovery of native species or ecosystems. If an invasive species is eradicated but the ecosystem becomes detrimentally affected by other erupting invasive species as a result of the eradication, the conservation programme should not be defined as a success. In other words, a programme cannot be qualified as a success if the proximate goal is reached (one management action) but the ultimate goal is not (species conservation). Eradication planning must therefore consider entire ecosystems and include assessments of the state of invaded ecosystems before drastic interventions such as the removal of deleterious invasive species (Thomas and Willis 1998). This step provides an estimation of the impacts of the invading species and enables predictions of the outcomes once eradication is completed. Such risk assessments need not be as detailed as ours for Surprise Island, but do require measures of the potential for other problematic alien invasive species to respond, so that, if necessary, they can be eradicated together, thus avoiding potential surprise effects such as chain reactions (e.g., Zavaleta *et al*. 2001). It is also necessary to implement the best control strategies qualitatively as well as quantitatively (Choquenot and Parkes 2001), according to local conditions. Of course, despite extensive study there can still be unexpected increases of invasive species following an eradication, but still with overall benefits to the natural ecosystem. In these cases, the eradication can be viewed as a success despite this surprise effect (Watari *et al*. 2011).

Sometimes, the risk of triggering a surprise effect might be worth taking in order to remove greater threats from particular invasive species. But when circumstances allow pre- and post- eradication surveys, the evidence gathered can provide lessons for other conservation programmes, help protect other ecosystems from invasions, and in the long run save money. Furthermore, scientific progress can be made out of what are essentially extraordinary situations. Biological invasions and alien species removals can both be viewed by theoretical ecologists as large scale experiments of trophic chain manipulations. Just as conservation practice has gained much from theoretical developments over the years, conservation biology can now be of tremendous help for fundamental ecology.

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Vegetation change following rabbit eradication on Lehua Island, Hawaiian Islands

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Abstract Lehua Island is a 112 ha tuff crater that lies 1.2 km north of Niihau in the Hawaiian Islands. This island hosts ten species of nesting seabird and is the second largest seabird colony in the main Hawaiian Islands. The first biological survey of the island was conducted in 1935 when European rabbits (*Oryctolagus cuniculus*) were already established on the island. A few years later Pacific rats (*Rattus exulans*) were detected. There are no historical data to indicate when these species were introduced and their impacts on the island community remain largely unknown. Fossil pollen and seabird bones indicate that there have been remarkable changes on the island. In an attempt to restore the island, rabbits were eradicated in November 2005. A series of transects were used to monitor vegetation change three years before and three years after rabbit eradication. A significant increase in vegetation cover and species diversity was immediately evident after rabbit eradication. The majority of this new biomass was largely due to the spectacular spread of the introduced grass *Setaria verticillata* in addition to introduced shrubs. Although plant cover of some native species increased, overall native plant cover declined. Of concern was the establishment of *Verbesina encelioides* (Asteraceae). This non-native herb has caused devastating changes to native plant communities and seabird habitat elsewhere in Hawaii. Rabbit removal in isolation does not appear to be beneficial for the conservation of Lehua Island. In such a highly invaded system, a successful restoration programme should incorporate concurrent control of non-native plants and non-native herbivores.

Keywords: Invasive species eradication, island restoration, *Oryctolagus cuniculus*, *Rattus exulans*, herbivory, introduced species, secondary effect

INTRODUCTION

The accidental or intentional introduction of alien species is one of the most serious threats faced by island ecosystems (Vitousek 1988). Rabbits (*Oryctolagus* sp.) have been released on more than 800 islands worldwide (Flux and Fullager 1983), often with devastating consequences. For example, rabbits introduced to Laysan Island in the Northwest Hawaiian Islands denuded the island within 20 years resulting in the loss of 22 species of plants and the extinction of three species of endemic landbirds (Watson 1961).

Introduced mammalian herbivores affect ecosystems in several ways. Through browsing, grazing, and trampling, they may cause the population decline of palatable plant species by decreasing their survival, growth, or fitness (Crawley 1997; Chapuis *et al*. 2004). At the community level, these effects can lead to drastic changes in diversity and species composition (Gilham 1961; North *et al*. 1994). Herbivore actions can lead to extensive erosion (Watson 1961; Kessler 2002) and stimulate cascading changes in entire ecosystems (Holmgren 2002; Maron *et al*. 2006).

Introduced herbivores can have devastating effects on those island plant communities without a history of vertebrate herbivory. Plants evolve defences in direct proportion to the risk of herbivory, and because defences are costly, production decreases when herbivore pressure is absent (Marquis 1991). Consequently, insular endemic plants that evolved in the absence of vertebrate herbivores typically lack defences against herbivory making them more palatable and susceptible to extirpation (Bowen and Van Vuren 1997).

Recent advances in techniques for removing introduced mammals from islands have made it an increasingly used management option. However, research has shown that species removal in isolation can also result in unexpected changes to other ecosystem components (North *et al*. 1994; Courchamp *et al*. 1999; Bergstrom *et al*. 2009). Furthermore, the secondary effects of alien removal become more likely as the number of interacting invaders increases in ecosystems and as aliens in late stages of invasion assume the functional roles of native species (Zavaleta *et al*. 2001).

Lehua Island is considered a priority site for conservation work by the Offshore Islet Restoration Committee (OIRC), which aims to preserve and restore Hawaiian offshore islets. There are no native terrestrial mammals presently or historically on the island, but two non-native mammals have been introduced. European rabbits (*Oryctolagus cuniculus*) were detected during the first survey of the island's flora and fauna in 1936 (Caum 1936) and the Coast Guard reported Pacific rats (*Rattus exulans*) in 1940 (Bishop Museum, vertebrate collection). The OIRC planned for the eradication of all non-native mammals from the island starting with a rabbit eradication programme in November 2005.

There is little historical data on the long-term effects of rats and rabbits on the Lehua island community. However, paleoecological studies indicate that there have been major changes on the island. Fossil pollen types identified on Lehua are typical of dry lowland forests, among the most endangered of all ecosystems in the Hawaiian archipelago. The following tree and shrub genera have been identified: *Psydrax, Pritchardia, Cordia, Thespesia, Rauvolfia, Zanthoxylum, Pittosporum, Dodonaea* and *Chenopodium* (OIRC, unpublished data). This contrasts with grassland/shrubland that was described during the first botanical survey of the island in 1936 (Caum 1936). Further altering the system is the introduction of 28 non-native species, which have become naturalised during the past 70 years (ca 56 total species present) and form a dominant component of the grassland/shrubland (Wood *et al*. 2004).

The goals of this study were to use the rabbit eradication as an opportunity to evaluate the secondary effects of herbivore removal in a highly altered ecosystem and to aid managers by identifying early invasions of non-native plant species.

METHODS

Study site and history

Lehua Island is an uninhabited tuff crater 1.2 km north of Niihau and 31 km west of Kauai, Hawaii (22°01'N, 160°06'W). The crater is highly eroded and nearly half submerged, forming a steep, crescent-shaped island of 112 ha with a maximum elevation of 213 m (Palmer 1936). The environment is harsh with highly seasonal precipitation and intense solar radiation. Annual rainfall is less than 600 mm with the majority falling during intense winter storms (Giambelluca *et al*. 1986). Lehua is the second

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largest seabird colony in the main Hawaiian islands, with 10 species nesting in large numbers (VanderWerf *et al*. 2007) and is protected as part of the Hawaii State Seabird Sanctuary. Nutrient input by seabirds significantly enriches soils and plants on the island (unpublished data).

As part of a Lehua restoration plan, it was hoped that the combined removal of rats and rabbits would reduce soil erosion, encourage colonization by small, rare seabird species and allow for an extensive planting effort. The rabbit eradication programme began in November 2005. Approximately 95% of the rabbits were killed within the first 10 days of hunting and the remainder were eradicated in January 2006 (Island Conservation, unpublished data). Logistical difficulties delayed the rat eradication until 2008.

Pacific rats were present on Lehua throughout the study period. Surveys and incidental observations indicate that the rat population increased after rabbit eradication. A rodent survey conducted before rabbit eradication in April 2004 detected no rats or rat activity in 154 trap nights (R. Doratt, unpublished data). Surveys conducted after rabbit eradication in June and September 2007 detected 137 rats in 500 trap nights and 39 rats in 223 trap nights respectively (R. Doratt, unpublished data). Incidental observations are consistent with these results as rats were increasingly commonly seen after rabbit eradication. Rats were observed regularly during the day and night, whereas prior to eradication such observations were extremely rare.

Vegetation monitoring

Vegetation monitoring began in September 2003, three years before rabbit eradication (effectively December 2005) and continued twice annually until April 2008. Sampling periods corresponded with the end of the wet season in April or May and the end of the dry season in September or October. Sampling focused on the most accessible, vegetated portions of the island. On the inner crescent, seven 100 m transects were randomly established in an east-west direction following the contours of the crescent. On the outer crescent, 15 x 50 m transects were randomly established along the lower ridges in a north-south direction with 1-3 transects on each ridge.

Point-intercept sampling (Mueller-Dombois and Ellenberg 1974) was used along these transects to estimate plant cover and species diversity (inverse of Simpson's index) for each transect during the sampling period. Sampling points were monitored at 1 m intervals noting species present at each point. Transect ends were marked with steel rods fitted with a PVC pipe for greater visibility and recorded with GPS. Plant cover was estimated by dividing the number of targets "hit" by the number of potential targets. To evaluate relative changes in the abundance of individual species, growth forms (forb, grass, shrub) or status (native, non-native), 2 x 2 chi-square contingency tables were used. To assess whether rabbit eradication affected total plant cover or mean species diversity, two-sample T-tests were used on the combined pre-eradication and post-eradication data. Statistical analyses were calculated using Minitab 15.

RESULTS

Two months after rabbit eradication, heavy rain (over twice the historical average) fell on Lehua from February 2006 to April 2006 (Fig. 1). Vegetation sampling in April 2006 showed a 53% increase in vegetation cover and a 71% increase in species diversity from the previous sample in October 2005.

Rabbit eradication was followed by a 59.7% increase in vegetation cover ($t = 5.54$, $p < 0.001$; Table 1; Fig. 2) that resulted from significant increases in non-native grasses and shrubs (Table 1). Cover by grasses increased by 83.3% (Chi-square value = 455.5, $p < 0.001$), predominantly from a rapid expansion of *Setaria verticillata*, and shrubs increased by 79.0% (Chi-square value = 25.0, $p < 0.001$). There was no significant change in forb cover. Overall, there was a 112.8% increase (Chi-square value = 751.0, $p < 0.001$) in cover by non-native species compared with a 33.9% decrease in the cover of native species (Chi-square value = 62.5, $p < 0.001$).

Plant diversity increased by 31.7% after rabbit eradication ($t = 4.12$, $p < 0.001$; Fig. 2). Ten new species were recorded in the study area. One was an indigenous forb, *Solanum americanum*. The remainder were non-native forbs (*Bidens pilosa, Boerhavia coccinea*, *Chenopodium carinatum, Conyza bonariensis, Crotalaria pallida, Emilia fosbergii, Sonchus oleraceus*) and grasses (*Chloris barbata, Digitaria* spp*., Paspalum conjugatum*). Although not detected in the study area, *Verbesina encelioides* became locally abundant after rabbit eradication and has since spread to different parts of the island.

DISCUSSION

Vegetation change following rabbit eradication

Vegetation on Lehua responds to winter rains with increases in cover and diversity, followed by a period of senescence during the dry season. The period of high rainfall coupled with rabbit removal in 2005 may have synergistically facilitated vegetation change. However, the effects were not due to rainfall alone as the increased vegetation cover and diversity remained significantly higher once rainfall levels returned to normal (Fig. 1).

The removal of rabbits from Lehua was followed by a remarkable increase in vegetative cover and a corresponding decrease in bare ground. This may have positive effects

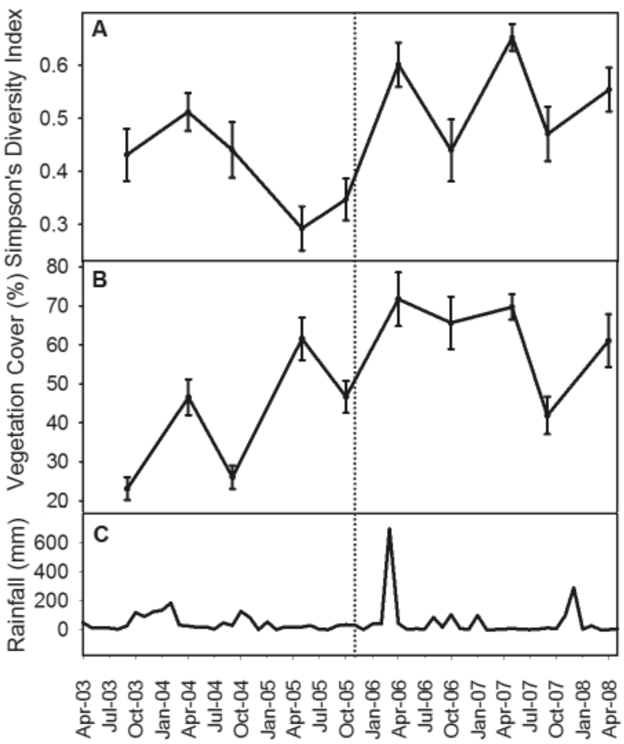

Fig. 1 A. Changes in species diversity with SE bars; B. Changes in vegetative cover with SE bars; and C. Rainfall between 2003 and 2008. The dashed line indicates when rabbits were effectively removed from Lehua. No weather station exists on Lehua. Monthly precipitation data were obtained from the closest weather station most resembling conditions on the island: Waimea rain gauge on Kauai's south shore (National Climatic Data Center).

Table 1 Frequency of occurrence and change in vegetative cover (%) after rabbit eradication. Chi-square analysis is not applicable to very small values; these species are indicated in gray. Native species are indicated with a (+). Bold text represents significant changes in vegetation cover when pre- and post-eradication of rabbits was compared.

Species	Frequency of occurrence pre eradication	Frequency of occurrence post eradication	Change in veg. cover (%)	Chi-square value
Grasses				
Cenchrus ciliaris	13.11	13.43	2.43	0.22
Cenchrus echinatus	0.37	1.66	**349.83**	41.98***
Chloris barbata	0.00	0.23	N/A	
Digitaria spp.	0.00	0.72	N/A	
Eragrostis amabilis	0.02	0.13	557.45	
Panicum torridum+	5.05	4.43	-12.34	2.11
Paspalum conjugatum	0.00	0.06	N/A	
Setaria verticillata	5.98	24.36	**307.35**	660.23***
Forbs				
Ageratum conyzoides	1.30	0.47	-64.02	19.04***
Bidens pilosa	0.00	0.02	N/A	
Conyza bonariensis	0.00	0.04	N/A	
Emilia fosbergii	0.00	0.09	N/A	
Gamochaeta purpurea	0.04	0.00	-100.00	
Sonchus oleraceus	0.02	0.02	9.57	
Chenopodium carinatum	0.00	0.19	N/A	
Jacquemontia ovalifolia+	10.33	5.57	**-46.04**	74.99***
Sicyos maximowiczii+	0.31	0.83	**167.09**	11.93***
Chamaesyce hirta	0.02	0.04	119.15	
Crotalaria pallida	0.00	0.02	N/A	
Waltheria indica+	0.33	1.85	**460.76**	54.42***
Boerhavia coccinea	0.00	0.21	N/A	
Boerhavia repens+	0.04	0.06	64.36	
Portulaca oleracea	0.02	1.98	10090.43	
Portulaca pilosa	0.21	0.06	-70.12	
Anagallis arvensis	0.17	0.11	-39.13	
Solanum americanum+	0.00	0.13	N/A	
Unknown forb	0.06	0.02	-63.48	
Shrubs				
Pluchea carolinensis	0.06	0.11	82.62	
Pluchea indica	1.03	1.53	48.86	4.96*
Abutilon grandifolium	1.09	2.26	**107.41**	20.72***
Combined values				
Bare ground	60.74	39.11	**-35.61**	459.97***
Grass	24.52	44.96	**83.32**	455.45***
Forb	12.91	11.89	-7.89	2.35
Shrub	2.17	3.89	**79.04**	24.99***
Native species	16.06	10.62	**-33.88**	62.45***
Non-native species	23.55	50.13	**112.83**	751.08***

*$p < 0.05$; **$p < 0.01$; ***$p < 0.001$

on the ecosystem through decreased erosion and increased burrow stability for nesting seabirds. However, the increased plant cover came from the release of non-native plants, primarily grasses and shrubs. These changes may seem counterintuitive as grasses and shrubs dominated the vegetation community before rabbit removal, but there are two factors at work to determine the effects of herbivory on plant community composition and structure. Herbivores directly affect vegetation through 1) feeding selectivity, and 2) recovery capacity of plants fed upon (see review by Augustine and McNaughton 1998). As such, the non-native grasses and shrubs must have been highly palatable (shown by their increase after rabbit removal), but also highly tolerant of tissue loss relative to other species, allowing them to achieve dominance under browsing pressure.

Two species of native plants increased in abundance (*Sicyos maximowiczii* and *Waltheria indica*; both browsed by rabbits) after rabbit eradication, but there was an overall decline in native plant cover. The native species that declined were likely less palatable to rabbits (no evidence of browsing damage), giving these natives a competitive advantage compared to highly palatable species. This advantage allowed the natives to co-dominate with competitive, fast-growing grasses. When released from herbivory, the grasses increased in range and density to the exclusion of these formerly abundant natives. Furthermore, the grasses have formed impenetrable mats in some areas precluding the germination of additional species. Similar trends have been observed following rabbit eradications elsewhere (e.g., Chapuis *et al.* 2004; North *et al.* 1994).

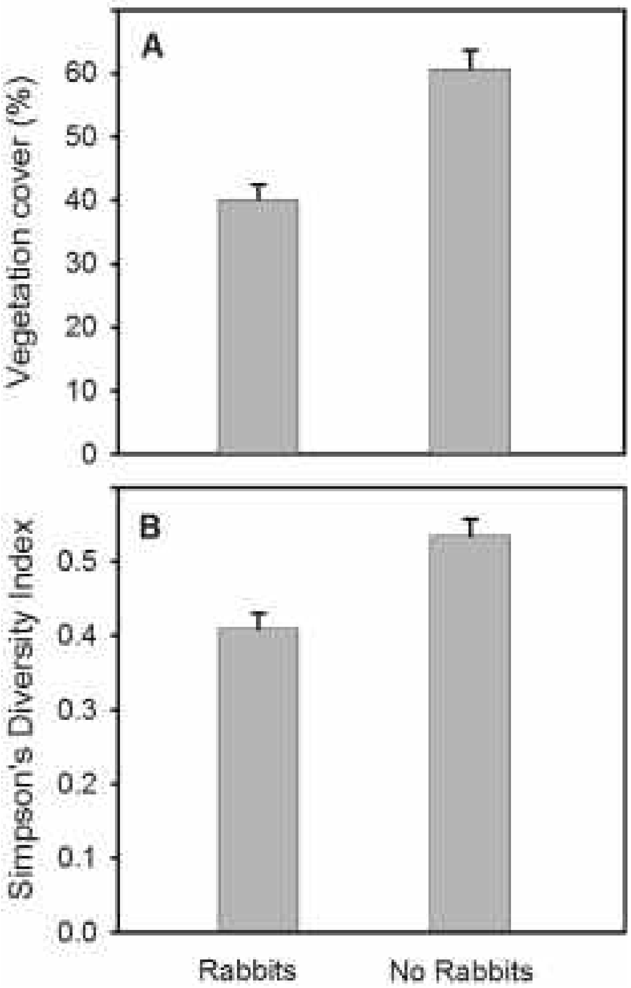

Fig. 2 Average change in vegetative cover A. and species diversity B. before and after rabbit eradication with SE bars. Both were statistically significant with $p < 0.001$.

Along with the increase in vegetation cover there was a significant increase in the mean number of plant species. This included ten species previously undetected in the study area, nine of which were non-native. Other incipient invasions of non-native species were found outside the study area including *Verbesina encelioides*, an invasive plant in the sunflower family that has caused devastating changes to native plant communities and seabird habitat elsewhere in Hawaii (Feenstra and Clements 2008). *Verbesina* has been reported on Lehua in the past but did not become established until rabbits were eradicated. This observation supports the theory that herbivores may suppress new plant invasions (Becerra and Bustamante 2007).

Rat influence

An attempted rat eradication, delayed until January 2008, was unsuccessful. After the rabbits were eradicated the rat population appeared to expand in response to an increase in available resources. Rats are omnivores that can alter vegetation composition, structure and dynamics through selective herbivory and granivory (Allen *et al.* 1994; Campbell and Atkinson 1999; Towns *et al.* 2006). They also contribute to recruitment depression through destruction of flowers, fruits, seeds, seedlings and plant parts (eg. Cuddihy and Stone 1990; Allen *et al.* 1994; Campbell and Atkinson 1999). This makes inferences about vegetation change following rabbit eradication challenging as herbivory by the increased rat population may be affecting plant community composition.

Effects of long-term herbivory

The premise behind the removal of rabbits and rats from Lehua Island was that because introduced herbivores target palatable species of plants their effects would be greatest on insular endemic species. In reality, the situation was more complex and involved links between introduced plants, rats and rabbits, plant palatability, tolerance to herbivory and competitive ability. Long-term suppression by herbivores, and subsequent depletion of the seed bank, results in a decline of preferred species which have low herbivory tolerance (Hunt 2001) and in some situations these changes can lead to alternate ecosystem states (Mack and D'Antonio 1998; Maron *et al.* 2006). Introduced rats and rabbits were present on Lehua for at least 70 years during which the ecosystem appears to have changed from a coastal dry forest to coastal dry grassland/shrubland. Of nine genera historically on the island, four are regarded as highly palatable to rats including *Pritchardia* (Athens *et al.* 2002; Perez *et al.* 2008), *Pittosporum* (Stone 1985; Cuddihy and Stone 1990), *Psydrax* (Medeiros *et al.* 1986) and *Zanthoxylum* (Cuddihy and Stone 1990). An additional genus is woody with large, fleshy fruits (*Rauvolfia*), which is also a characteristic favoured by rats (Meyer and Butaud

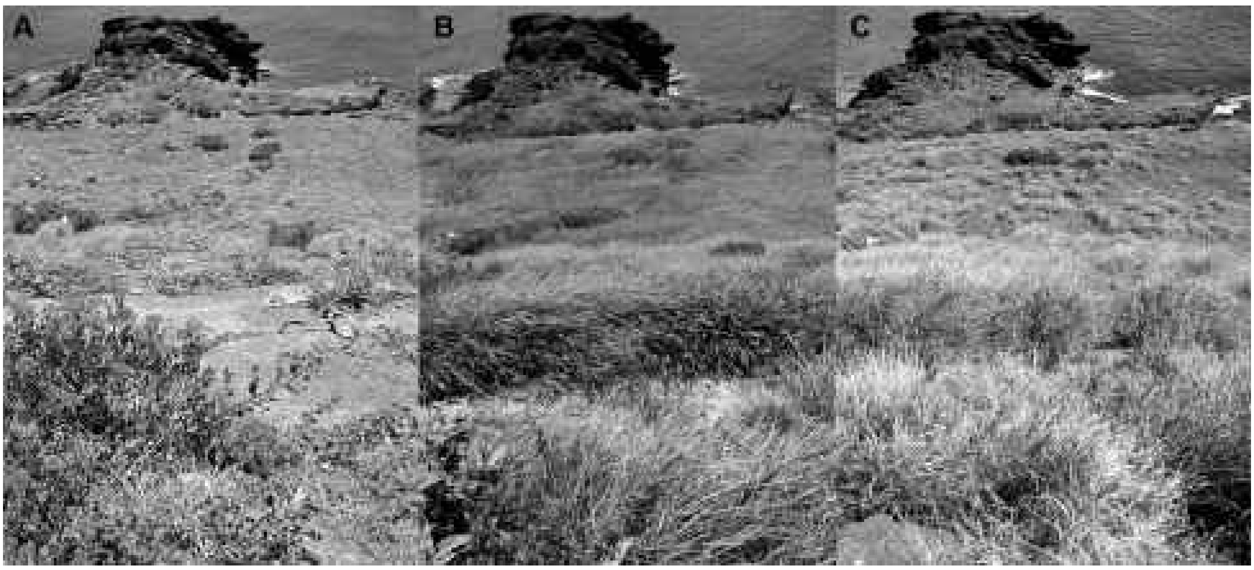

Fig. 3 Photo comparisons of transects on the outer crescent monitored in A. April 2004, before to rabbit eradication; B. April 2006, after rabbit eradication and an extremely wet season; and C. April 2008, more than two years after rabbit eradication.

2009). More recently, extirpated native species such as the succulents *Scaevola sericea* and *Portulaca villosa* may have been targeted by both rats and rabbits.

Conservation implications

The removal of non-native herbivores is presumed to have beneficial effects for native plant communities, especially since herbivore-induced changes can be reversible in some situations (eg. Copson and Whinam 1998; Donlan *et al*. 2002). However, when ecosystems experience multiple, or long-term invasions, the situation can become increasingly complicated and chances of successful reversal may be less likely (Zavaleta 2002; Courchamp *et al*., 2003). The adverse effects caused by the long-term presence of rabbits and rats on Lehua in combination with introduced plant species has resulted in a highly altered ecosystem. In this new system, introduced rabbits suppressed non-native plants and the removal of rabbits in isolation resulted in an increase of non-native plant cover, a corresponding decrease in native plant cover, and an increase in the abundance of rats. A problem for restoration of this seabird sanctuary is the increased rat population that followed rabbit removal. This may be increasing predation pressure on nesting birds. Additionally, short-term management of the island in the presence of rats means efforts to replace non-native plant species with native species must be delayed as native species are sensitive to rat damage. In situations like Lehua a restoration programme should address concurrent control of all non-native plants and animals.

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Live capture and removal of feral sheep from eastern Santa Cruz Island, California

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Abstract Sheep (*Ovis aries*) were brought to Santa Cruz Island, one of the eight California Channel Islands, in the mid-1800s. The islands were ranched throughout the 19th and most of the 20th century. Hunting of feral sheep occurred during the late 20th century. The Nature Conservancy (TNC) purchased the western 90% of the island in 1979 and eliminated sheep from their property by 1989. Feral sheep remained on the eastern 10% of Santa Cruz Island (ESCI) and supported a private sport hunting operation. The National Park Service (NPS) completed acquisition of ESCI in 1997. The sheep, private property of the former landowners, had to be either purchased or relocated. NPS opted to live capture and move the sheep to the mainland. It was thought there were approximately 2300 sheep at the time. Capture operations began in May 1997 using herding and corral traps with bait. As sheep capture became more difficult additional techniques were tested. Herding into corral traps in strategic locations was the most efficient technique. As numbers declined, sheep were individually pursued and captured. Transport of animals from the island involved loading sheep into stock trailers and driving the trailers onto a landing craft. The project was declared complete in December 1999 with over 9200 sheep captured. However, remnant sheep were found several times and the last sheep was removed in February 2001. Each of the California Channel Islands had sheep for some time during the ranching era of the 19th and 20th centuries. This project ended this chapter in the history of the Channel Islands.

Keywords: *Ovis aries*, California Channel Islands, eradication, island restoration.

INTRODUCTION

Santa Cruz Island, 25,000 ha (Fig. 1), is the largest of the five islands in Channel Island National Park off the coast of southern California The island has rugged terrain that reaches 747 m, steep canyons, extreme slopes and perennial and ephemeral streams. Climate is Mediterranean with plant communities predominantly grassland, island chaparral, island and southern coastal oak woodlands, bishop pine forest, and coastal-sage scrub (Minnich 1980; Junak *et al.* 1995).

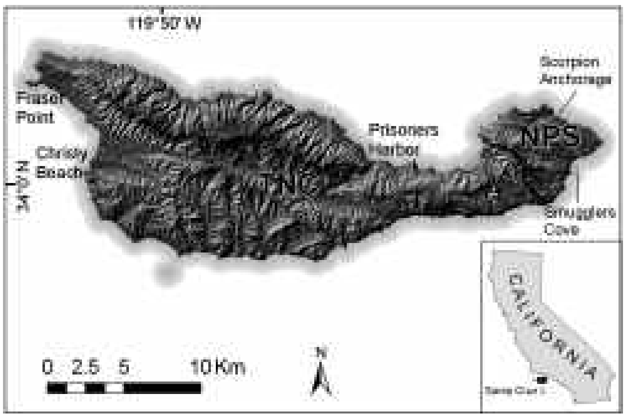

Fig. 1 Santa Cruz Island; the largest of the eight California Channel Islands.

Santa Cruz Island wss a Mexican land grant, which in the mid 1800s was transferred to private owners who shortly afterwards introduced sheep (*Ovis aries*) for wool and meat (US District Court 1857; Brumbaugh 1980). Periodic roundups of these Merino-Rambouillet-Churro sheep (Oklahoma State Univ. 1998; Van Vuren 1981) captured at least 50,000 (Towne and Wentworth 1945) (Fig. 2), which probably accounted for no more than half the population at any one time (Symmes and Associates 1922). Various fence lines and fenced pastures were constructed to facilitate the round-up. By the 1920s, sheep were essentially feral over extensive areas. In 1925, ownership was partitioned among descendants of the former landowner into two separate parcels. One parcel, forming the western 90% of the island (WSCI), was purchased by The Nature Conservancy (TNC) in 1979. The eastern 10% (ESCI) was purchased by the National Park Service (NPS) in 1997 through a "legislative taking" authorised by the U. S. Congress (Public Law 104-333; Sec. 817). Formal sheep ranching ended in the mid-1960s on WSCI and in 1984 on ESCI. By 1997, ESCI was primarily bare ground and overgrazed grassland, almost devoid of trees and severely eroded.

Fig. 2 Sheep round-up circa 1920 on Santa Cruz Island, California, USA.

Upon acquisition of WSCI, TNC determined that sheep were the main cause of habitat destruction and the greatest threat to the island's native biota (Brumbaugh 1980; Hochberg *et al*. 1980; Van Vuren 1981; Van Vuren and Coblentz 1987). Sheep also affected cultural and historic artifacts (Van Vuren 1982). TNC and volunteers began feral sheep eradication between 1981 and 1989 on most of WSCI and shot 37,717 sheep at a cost of US$240,000 (Schuyler 1987, 1993). TNC constructed a fence near their eastern boundary to restrict entry of sheep from ESCI, which left sheep on 3300 hectares of the island (this includes 800 ha of TNC property to the east of the sheep fence). The owners of ESCI at the time controlled the feral sheep (and feral pigs present throughout the island) with guided sport hunts for paying clients. This control ceased with the acquisition of ESCI by NPS.

The mandate of the NPS is to "Conserve the scenery, natural and historic objects and wild life therein and leave them unimpaired for future generations". Thus, the NPS decided to eliminate the remaining sheep from the island due to their ecological and archaeological impacts. Because

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the feral sheep were the property of the former landowners, the NPS had a legal responsibility to either purchase the animals or relocate them to the mainland. Animal rights organisations quickly announced that they would fight any project that involved killing the sheep. The NPS felt it would be possible to eliminate the sheep through live-capture and transport.

This paper outlines the methods developed over four years that enabled the eventual removal of all remaining sheep on ESCI through live capture and transport. We also describe some subsequent changes to the island ecosystems.

MATERIALS AND METHODS

Developmental phases

Upon acquisition of ESCI, the NPS immediately began developing an infrastructure that would support the removal of sheep including: 1) the rebuilding of corrals and fences, construction of walk-in traps; 2) purchase and modification of stock trailers; and 3) acquisition of baits, all terrain vehicles (ATVs), and sheep-holding containers for slinging under a helicopter. Fencing consisted of standard 1.2 metre high wire sheep fences hung on 1.8 metre metal T-posts. A veterinarian provided extensive advice and review of planned procedures for the holding and transporting of sheep.

The first year of the sheep capture and removal (Phase I; Table 1) was carried out by NPS personnel, including two Navajo sheep herders hired specifically for the project. The infrastructure to hold, feed, and transport the sheep was also developed during this period. NPS operations began by capturing sheep in the most accessible pastures by baiting into corrals using water, apple mash, molasses, hay, and sweet feed. One attempt was made to herd sheep using a line of 32 people and a helicopter.

In early 1998, the NPS solicited contractors with sheep herding experience to assist with the project. Phase II of the project began in June 1998 by contracting a sheep herding company with border collie dogs trained to work with large flocks of domestic sheep. However, this contractor worked for less than one month. Phase III involved the NPS continuing capture of sheep while searching for a new contractor with the necessary skills to capture the many feral sheep remaining on ESCI.

In Phase IV, over 70% of the sheep were captured with assistance from a contractor who specialised in the capture of livestock such as feral cattle, using dogs and people on horseback. Once it was thought that all sheep had been captured and removed, NPS staff on the island continued to monitor backcountry areas for sign of sheep.

Aerial surveys

Surveys using a Bell Jet Ranger helicopter flying along the contours of ESCI were conducted in the early morning of 12 and 13 November 1998 using three and two observers respectively. Each survey took approximately two hours to cover the approximately 3300 ha potentially occupied by sheep. More than 5000 sheep had already been removed and the primary purpose of the surveys was to estimate the number and distribution of sheep remaining.

Capture and transport

Most sheep were caught by herding them into large corral traps in strategic locations. The sheep were moved by a combination of personnel on foot, horses, and ATVs, and with the help of dogs and occasionally a helicopter. Passive baiting to attract sheep to corrals was used in Phase I of the programme when animals were particularly numerous and in need of food or water. Other methods attempted but less effective than herding were net gunning, drive-nets, boma fence, darting, pop-up traps, cannon nets, drop nets, night capture, plastic barriers, noise makers, plastic fencing, and telemetry. When sheep numbers were reduced, individual animals were pursued and captured.

Upon capture, sheep were transported by herding, vehicle, or helicopter, held in corrals near Scorpion Harbor, and shipped to the mainland. The holding pens, along with lanes and gates, were built in a sheltered area to separate sheep and facilitate loading operations. Food and water were provided as some sheep were kept in the pen for a week or more. Sheep were generally held until numbers reached approximately 200 animals, which justified running a landing craft.

Sheep about to be shipped were loaded into 6.7 m stock trailers modified with a centre platform to create a top and bottom that could be sectioned into four compartments. Each trailer could carry 75 sheep and was pulled by a ¾ ton pickup truck. Up to three trailers were loaded onto the park's 24 m landing craft for the 35 km trip across the channel to the Port of Hueneme on the California mainland. Tractors and forklifts were used to tow trailers onto and off the landing craft. The trailers were offloaded at the mainland and driven 145 km north to a stockyard in Buellton, California to be transferred to their owners.

Monitoring

Post-project monitoring was informal and carried out primarily by park and TNC personnel in the course of their other duties on the island. Staff carried out regular surveys by foot or vehicle throughout ESCI. Efforts concentrated on preferred habitats, such as water sources and canyons.

RESULTS

Size of sheep population

It is unknown how many sheep were on ESCI at the time of acquisition by NPS. Densities on highly impacted areas on WSCI were estimated at approximately 2 sheep/ha (Van Vuren 1981; Van Vuren and Coblentz 1987; Schuyler 1993). A similar density on ESCI (including the additional 800 ha of TNC property to the east of the sheep boundary fence) would indicate approximately 6600 sheep. However, in the seven months prior to NPS acquisition, the landowners had shot approximately 3000 sheep as part of their sport hunting operation. The hunt operators estimated that 2300 sheep were on ESCI at the time of the land purchase and the NPS began the project believing this to be a good estimate of numbers.

Table 1 Capture of sheep.

Period	Phase	Number of sheep captured
May 1997 – May 1998	I. NPS Initial Operations	1999
June 1998	II. Contractor #1	328
June 1998 – Sept. 1998	III. NPS Operations	273
August 1998 – December 1999	IV. Contractor #2 (Lausten) and NPS Operations	6653 (3822 of these following the Nov. 1998 aerial count)
January 2000 – February 2001	V. Monitoring and removal of remnant sheep	6
TOTAL		9259

Another estimate can be made by back-calculating from the number of sheep captured and using the following assumptions about feral sheep: 1) female to male ratio is 1:1, and 2) the productivity of females is almost 100% (Van Vuren 1981; Van Vuren and Coblentz 1987; Griffin 1976). This suggests annual recruitment of between 40%-50% of the total population. While lambs were found at any time of the year, most reproduction was during winter (Van Vuren and Coblentz 1987). Assuming 45% annual recruitment to the total population, this amounted to a monthly increase of 3.8%. Monthly capture rates were calculated by dividing the total captures for a period by the months of that period. This resulted in an estimation of roughly 5000 sheep present on ESCI at the beginning of the project; more than double the belief of the former hunting guides.

Aerial counts

The aerial counts in November 1998 tallied 1889 and 1712 sheep, respectively. Observers estimated they had likely missed approximately 15% of the animals and therefore thought the number of sheep was likely approximately 2100 animals. Most aerial counts have been shown to be underestimates (Caughley 1974; Cook and Jacobson 1979; Caughely and Grice 1982; Gasaway *et al.*1986; Pollock and Kendall 1987), and the proportion that escapes detection can be up to 50% of the animals (Stoll *et al.* 1991).

In the year following the count (between December 1998 and December 1999) 3822 sheep were removed. Assuming an annual increase of 45 %, and that lambing was shortly after the count, the population during the survey would have been 2,635 animals; an under-estimation by the aerial count of up to 35%. However, since ESCI was relatively barren of trees and with woody vegetation cover of appeared to be less than 10%, a 35% undercount is much higher than estimated by observers. Sheep may have been able to hide in ravines and on cliff-faces. Furthermore, the earlier use of the helicopter for herding may have increased aversion behaviour by the sheep

Capture of sheep

In the first year of operation, NPS staff used the various herding methods to capture and relocate nearly 2000 sheep from the island. Initially, the extremely poor condition of habitats due to an unsustainable number of sheep resulted in large numbers of animals voluntarily entering corrals in search of water. As sheep numbers declined and food became more available, capture operations shifted to herding or pursuing individual animals. Herding was useful for removing sheep, but not at a rate that could keep up with recruitment through births.

In December 1997, a major rainstorm and subsequent flood damage halted capture of sheep and diverted approximately six months of work to repairing of housing, fences, and facilities. The rains also created ideal conditions for vegetation regrowth and likely resulted in substantial recovery of the sheep population.

In June 1998, the first of two contractors began the planned use of eight sheep herding dogs (border collies) and four personnel to herd sheep into corral traps. Hundreds of sheep were moved into traps but their pace of entry outstripped the staff, the trap gate was not closed in time, and all sheep escaped. Having experienced this trap, no sheep would re-enter it. Although morale shattering for those involved, there were lessons learned: 1) a method of rapidly closing the gate was needed, and 2) sheep will learn to avoid traps. Camouflaging someone near the gate was tried, but the sheep could detect their close proximity through scent and would become skittish and suspicious. The border collies worked well at first, but the sheep learned that they could bolt past the dogs by exploiting the extreme terrain and their overwhelming numbers. The contractor left after two weeks having captured 328 sheep.

A single attempt to drive sheep with a line of people and a helicopter resulted in the capture of only one sheep. This sheep drive initially moved many hundreds of sheep. However, all but one of the animals eventually ran around or through the line.

These problems were resolved by engaging a “cowboy” livestock company, Ralph Lausten, Inc., experienced with horses and cattle dogs, which coordinated with NPS personnel. People on foot, horseback, and ATVs herded the sheep into traps, exploiting the terrain and using people on horses to quickly close a distant corral gate. Traps and fence lines were then inserted into each section of the island that constituted a flock’s home range. The sheep’s inability to easily migrate into a new area was exploited by clearing sections as rapidly as possible. This allowed sheep to be cleared from a section without educating the adjoining flocks. Stragglers were left to be dealt with later.

In this way, the island was divided into sections that were quickly cleared of most sheep. By December 1999, all sheep had been captured except for a few isolated individuals, which were removed by the park personnel when discovered. The last sheep was found hiding in a heavily vegetated area on TNC property to the west of the sheep boundary fence.

Between 1997 and 2001, 9259 sheep were removed from Santa Cruz Island (Table 1). One animal died in transport between the island and the sheep yard in Buellton, California. It is not known how many animals died or were injured during capture or holding on the island. After 9253 sheep had been captured park staff and the contractor believed that all sheep had been eliminated by December 1999. However, in November 2000, TNC reported sheep on their property. Over the following two months, six sheep were located and removed. The last of the sheep was found in February 2001. The total cost for capture and relocation of the sheep to the mainland was approximately US$2,000,000 (J. Fitzgerald, Channel Islands National Park; pers. comm.).

DISCUSSION

Population size and monitoring

Santa Cruz Island is now free of sheep after 150 years of their effects on landscapes and native vegetation. This was only achieved after some hard lessons were learned. The first of these was that estimations of the size of animal populations vary greatly when “gut feelings” are used rather than structured surveys. Additionally, the estimated near doubling of sheep numbers, from 5000 to 9259 animals during the three years of the project, illustrates the substantially increased productivity as food resources improved. This highlights the need for sufficient resources to complete removal projects as quickly as possible. For this well-funded project, vague estimates of population sizes did not alter the outcome. However, large underestimates of the number of animals could be the difference between success and failure for many projects unable to sustain necessary funding or management support.

Over a year was required to detect and remove the last sheep, which demonstrates the necessity of monitoring after such a project. Monitoring can be the most expensive aspect of a project with little to show for the funds expended. However, it must be planned for and resources set aside in order to properly conduct searches for the last animals. We recommend that projects to remove feral animals commit much greater resources to monitoring than was done in this case.

Unexpected problem with sheep behaviour

In addition to difficulties with herding sheep using dogs, flocking behaviour by sheep was initially a hindrance, but one that became an advantage. Staff initially attempted but failed to herd sheep out of their home range and into corrals in another area. Boyd (1981) remarked that feral sheep on St. Kilda "…formed a close flock when disturbed, running to the limits of their home range before doubling back". This describes our experiences regarding sheep behaviour. On Santa Cruz Island, sheep had home ranges from 20 to 300 ha. Rams covered a greater area then ewes. The sheep expanded their home range in the fall and winter when vegetation was scarcest (Van Vuren 1981). Our initial lack of understanding of sheep home range resulted in expenditure of time and resources for no gain. Once this aspect of behaviour was understood, it was exploited and used to section the project area into management units. Corrals were then built in each home range unit and we did not attempt to move sheep out of their range.

Cost

The NPS spent approximately US$2,000,000 to live capture and transport 9259 sheep to the mainland; a cost of US$216/sheep. By comparison, TNC spent approximately US$240,000 to eliminate 37,717 sheep on their property between 1981 and 1989 (Schuyler 1987). To compare the cost/sheep between the NPS and TNC projects, we used 1985 as a midpoint for the TNC project and adjusted for inflation. The estimated cost of the NPS project in 1999 was US$371,000; or about US$10/sheep. In addition, the NPS would have had to pay the sheep owners an unknown fair market value for their animals.

Island projects tend to have higher costs than similar mainland projects because of the need to transport people, equipment, and supplies by boat or air to the island. Since both projects were done on the same island, there are many similarities in the logistical and environmental challenges and costs. Part of the explanation of cost difference is that TNC used volunteers extensively for their project, while all of the workers on the NPS project were paid staff or contractors. However, the primary explanation for the cost difference is that live capture and transport of the sheep is inherently more expensive than direct reduction. Of the US$216/sheep cost, approximately US$60/sheep was spent to transport animals from the island to the Buellton stockyard. This cost did not vary much through the project. However, the cost to capture each sheep increased greatly as the project progressed and more expensive methods were used. In the last year of the project a helicopter was used extensively for locating remnant sheep and for transporting sheep in a cage slung from the helicopter.

Additional costs included the construction and maintenance of temporary infrastructure (fencelines, traps, corrals), acquisition of support equipment (sheep trailers), and the care and feeding of sheep. Finally, the extended duration of the project, resulted in the handling of greatly increased numbers of sheep.

Recovery of island ecosystem

The primary reason for removing sheep from Santa Cruz Island was to protect and restore the unique island ecosystem. The island, never connected to mainland California, provides habitat for over 600 species of vascular plants including at least 8 endemic taxa (Junak *et al.* 1995), one species of endemic snake, and four species of endemic mammals (Schoenherr *et al.* 1999). There is also an endemic species of bird, the island scrub jay (*Aphelocoma insularis*). Island scrub jays, which prefer oak woodland and chaparral habitat, are currently uncommon on ESCI.

The removal of feral sheep from the TNC property in the late 1980s resulted in dramatic and rapid changes in the soils and vegetation. As vegetation began recovering on TNC property, differences in vegetative cover between the western and eastern portions of the island developed. The boundary fence between the properties delimited recovering vegetation and bare ground that was visible (Fig. 3). The demarcation was even visible in satellite photos. As the vegetation recovery on ESCI progresses, the line has become less dramatic.

The difference in timing for sheep removal from TNC and NPS property provided an opportunity to assess the impact of sheep on frequency of landslides. Widespread slope failures were highly correlated with the presence of sheep. In the 1970s, slope failures were common over the entire island (Pinter and Vestal 2005). By the late 1990s, 80% of slides were on the 10% of the island with sheep (Pinter and Vestal 2005). Within four years of the removal of sheep from ESCI, vegetation recovery there was sufficient to substantially reduce slope failures in spite of heavy rains during the winter of 2004-2005 (Pinter and Vestal 2005).

The removal of grazers is allowing the expression of some aggressive non-native plants that have the potential to dominate vegetation communities. NPS staff are controlling high priority plant species. Olive (*Olea europaea*) seedlings, originating from planted groves on ESCI, virtually exploded throughout the project area; between 2005 and 2009, park staff removed more than 11,000 plants (P. Power, Channel Islands National Park; pers comm.). If not controlled, feral olives threaten the recovering native plant communities and have the potential to transform the native shrub and grassland communities to non-native woodland.

There has been a substantial increased in vegetation cover over the whole island but most of the vegetative on ESCI continues to of be non-native species (Klinger *et al.* 2002; Morrison 2007). Although ESCI lags behind the TNC property in the recovery of trees, shrubs, and other native plants, it is beginning to show decreased cover of bare ground, increasing herbaceous cover, and growth of native woody plants. NPS and TNC are continuing to monitor and assess invasive plant species and prioritise control activities.

In 1994, nine endemic plant species were federally listed as threatened or endangered on Santa Cruz Island. Habitat alteration and soil loss were identified as threats to recovery of all of the listed species (US Fish and Wildlife Service 2000). The last known location of *Malacothrix squalida*, an endangered annual plant, had been on ESCI in the 1960s (S. Junak, Santa Barbara Botanic Garden; *pers. comm.*). However, since the removal of the sheep, it is being seen again on ESCI.

Fig. 3 Sheep fence clearly showing the effects of sheep on the right contrasted with no sheep for approximately 15 years on the left. Santa Cruz Island, California, USA.

Native animals are also expected to respond positively to the removal of feral sheep. Drost *et al*. (2009) found that Santa Cruz Island deer mouse (*Peromyscus maniculatus santacruzae*), and Santa Cruz Island harvest mouse (*Reithrodontomys megalotis santacruzae*) have increased in numbers and the harvest mouse has increased in distribution. It is likely that the improved food and cover resulting from sheep removal is supporting increases in mouse populations.

Feral pigs (*Sus scrofa*) were also present on Santa Cruz Island and impacted soils and vegetation. Pigs were the last species of non-native mammals on the island, and were eliminated between 2005 and 2006 under a programme carried out jointly by NPS, TNC, and contractor Prohunt, Inc (Parkes *et al.* 2010). The elimination of the feral pigs closed the approximately 150 year chapter of the island's ranching history. The inclusion of Santa Cruz Island into Channel Islands National Park in 1980 and the acquisition of the island by TNC and NPS represented a major shift in the purposes for which the island is valued by the public. We are now in a period of ecological restoration. The island ecosystem will continue to face many threats. However, it is hoped that a more intact and resilient ecosystem will allow the many unique taxa and ecosystem processes to persist long into the future.

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Environmental monitoring for brodifacoum residues after aerial application of baits for rodent eradication

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Abstract Aerial application of brodifacoum bait for eradication of invasive rodents from islands raises concerns about environmental contamination and adverse effects on non-target wildlife. We summarise results of environmental monitoring for brodifacoum residues after New Zealand eradications in a fenced reserve at Maungatautari and on the offshore islands Little Barrier, Rangitoto and Motutapu. Brodifacoum was not detected in extensive fresh water monitoring at Maungatautari, or in fresh water samples from Little Barrier Island. Residual concentrations were present in soil samples from underneath degrading bait pellets on Little Barrier, and decreased to near the limit of detection by *C.* 100 days after application. No brodifacoum was detected in marine shellfish sampled from Little Barrier, Rangitoto or Motutapu. A range of birds, including a kiwi from Little Barrier, were considered non-target mortalities. Residual brodifacoum concentrations detected in three of nine little blue penguins found dead on beaches outside the Rangitoto/ Motutapu area after baiting were considered to represent sublethal exposure, with starvation considered the likely cause of death. This result highlights the crucial role of post-application environmental monitoring in rodent eradications, in addressing community concerns and filling information gaps regarding the movement, persistence and effects of brodifacoum in the environment.

Keywords: Laboratory testing, non-target species, residues, rodenticides, soil, water

INTRODUCTION

Brodifacoum is among the most toxic of the anticoagulants used against rats and mice (Erickson and Urban 2004), so they need to ingest a relatively small amount of bait for lethal exposure. The product has become a valuable tool for island conservation because of its delayed toxicity (Kaukeinen and Rampaud 1986), high rodenticidal efficacy, and bait formulations that are highly acceptable to rodents and can be applied aerially over large areas. To date, brodifacoum baiting has been used in an estimated 71% of campaigns to eradicate introduced rodents from islands (Howald *et al*. 2007). An important consideration has been assessing risk to non-target wildlife and the potential for environmental contamination. Increasingly, rodent eradication is being considered for islands that are inhabited or used by people or are close to highly populated mainland areas. Where the use of brodifacoum bait is proposed, particularly through aerial application, managers also need to address possible environmental contamination pathways that pose risks to humans, livestock and domestic animals.

Here we describe monitoring undertaken after three New Zealand eradications of rodents from islands involving aerial application of cereal pellet bait containing 20 ppm brodifacoum. We discuss the results in the context of environmental contamination and non-target risk. Under current New Zealand legislation, the aerial discharge of a contaminant such as brodifacoum to land and water (e.g., using helicopters for bait application) requires consent from a local government agency. While there are currently no prescriptive environmental monitoring regimes for residual brodifacoum, concerns addressed during the consent application process for each of the eradications focused attention on the fate of brodifacoum in water and soil as potential transfer pathways to human food and non-target wildlife. Where aerial application could result in bait entering the marine environment this has included monitoring of coastal marine fauna, especially shellfish commonly harvested for human food.

METHODS

Maungatautari water monitoring

The Maungatautari Ecological Island Trust (MEIT) aims to achieve complete pest mammal eradication in this mainland reserve in the central North Island, by pest-proof fencing and removal of pest mammals through aerial baiting and trapping within the fenced area (see www.maungatrust.org/index.asp). A pilot eradication programme in two fenced enclosures on the northern (*c.* 32 ha) and southern (*c.* 76 ha) sides of the mountain was undertaken in 2004. Each enclosure received two aerial applications of Pestoff Rodent Bait 20R at a rate of 15 kg/ha, applied in accordance with a Code of Practice (Anon. 2006). Streams flowing through both enclosures were used for human or livestock drinking supply by adjoining landowners. The resource consent specified that all water supplies drawn from the enclosures be disconnected before bait application, and to remain so until two water samples taken on consecutive days showed that any brodifacoum contamination was below the analytical method detection limit (MDL).

Samples from two streams in each enclosure were taken at zero hours (baseline) then at 1, 2, 3, 6, 9, 12, 24, 48 and 72 h after bait application, and thereafter at intervals of one week, two weeks and three months. Further samples were taken after ≥25 mm rainfall occurred in a 24-h period. Samples were taken from the point where each stream left the enclosure and at *C.* 800 m downstream. Samples taken up to 48 h after bait application were analysed within 24 h of receipt by the laboratory, to facilitate reconnection of water supplies once there were two consecutive below-MDL results.

Little Barrier Island water, soil, bait degradation and marine shellfish monitoring

Little Barrier Island is in the Hauraki Gulf 80 km north-east of Auckland (see www.doc.govt.nz/parks-and-recreation/places-to-visit/auckland/warkworth-area/little-barrier-island-hauturu-nature-reserve/). The Department of Conservation (DOC) aerially spread Pestoff Rodent Bait 20R at 11.7 and 6.2 kg/ha in June and July 2004, and the island was declared free of Pacific rats (*Rattus exulans*) in July 2006.

Carcass searches along the island's track network and grid-searches over *C.* 120 ha were undertaken during the week following each bait application. One kiwi carcass recovered was necropsied (IVABS, Massey University, NZ) with liver tissue analysed for residual brodifacoum (Table 1). Monitoring of bait degradation was used to

Table 1 Testing laboratories, numbers analysed and detection limits for water, soil and animal tissue samples tested for residual brodifacoum following aerial bait application.

Island eradication	Sample type	No. tested	Testing laboratory	MDL (ppm)
Maungatautari	Water	217	LCR	0.00002
Little Barrier	Water	4	AQ	not specified
	Soil	8	AQ	not specified
	Shellfish	4*	AQ	0.001
	Kiwi liver	1	LCR	0.001
Rangitoto/Motutapu	Water	4	LCR	0.00002
	Shellfish	2*	LCR	0.001
	Penguin liver	9	LCR	0.001
	Dolphin liver	5	AQ	0.005
	Dolphin ingesta	5	AQ	0.005
	Dog vomit	1	AQ	0.005
	Pilchards	1*	LCR	0.001

* Each sample consisted of four or five individual shell/fish combined.
LCR = Landcare Research Toxicology Laboratory, Lincoln, New Zealand. AQ = Agriquality National Chemical Residue Laboratory, Upper Hutt, New Zealand. MDL = method detection limit.

determine timing of the release of three brown teal (*Anas chlorotis*) taken into captivity before the operation. At four sites representing grassland and forested habitats across the island, 20 bait pellets were placed under wire cages designed to exclude rodents and birds, and checked for condition scoring following the categories described by Craddock (2003a), over four months. Soil monitoring was undertaken after peg-marking the position of individual pellets so that soil samples could later be taken from the exact location. Soil (4-cm^3 plugs) collected at days 56 and 153 after the second bait application was stored frozen until analysis. Within 24 h after both bait applications, water samples were taken from one waterway, less than 1 m downstream from where bait pellets were visible in the water, and also from the island's bore water supply. At one and two weeks after the second bait application, samples (Table 1) of paua (*Haliotis iris*) and scallops (*Pecten novaezelandiae*) were taken from within 5 and 50 m of the shoreline, respectively.

Rangitoto and Motutapu islands residues in water, wildlife and marine shellfish

Rangitoto and Motutapu are connected islands in the inner Hauraki Gulf, approximately 8 km north-east of Auckland (see www.doc.govt.nz/parks-and-recreation/places-to-visit/auckland/auckland-area/rangitoto-island-scenic-reserve/ and www.doc.govt.nz/parks-and-recreation/places-to-visit/auckland/auckland-area/motutapu-island-recreation-reserve/). DOC undertook three aerial applications of Pestoff Rodent Bait 20R on 19-20 June, 9 July and 6 August 2009 with respective application rates of 22.1, 9.5 and 6.6 kg/ha. The initial high application rate was used to minimise the risk that uptake by rabbits would leave gaps in bait coverage intended for rodents (*Rattus rattus*, *R. norvegicus* and *Mus musculus*). Roof water-collection systems were disconnected before aerial application, and roofs and animal drinking troughs cleared of any bait afterwards. Four samples from drinking supplies on Motutapu were taken approximately 2 months after the last aerial application. Three weeks after the last application, 10 pipi (*Paphies australe*) from Motutapu and 10 mussels (*Mytilus edulis*) from Rangitoto were sampled for residue testing (Table 1).

The weeks following the baiting operation coincided with cases of domestic dogs (*Canis familiaris*) being poisoned on Auckland beaches. A vomit sample from one of five dogs that died was tested for residual brodifacoum (Table 1), although veterinary diagnoses and chemical testing later indicated that these cases were the result of dogs ingesting sea slugs (*Pleurobranchaea maculata*) containing tetrodotoxin (McNabb *et al*. 2009). The death of dogs soon after the Rangitoto/Motutapu brodifacoum applications increased public awareness of the aerial application of brodifacoum. National media and Internet coverage was given to assertions by various interest groups and individuals that marine wildlife, including little blue penguins (*Eudyptula minor*), dolphins (*Delphinus* sp.) and pilchards (*Sarditlops neopilchardus*), found dead on local beaches outside the eradication operational area had been poisoned as a result of the eradication operation. To address these concerns, brodifacoum testing was carried out on samples of liver from nine little blue penguins, samples of dolphins' stomach contents and samples of whole pilchards (Table 1). Necropsy data was also obtained to further diagnose whether brodifacoum poisoning was likely in these cases.

Residue analyses

Two accredited New Zealand laboratories analysed samples for brodifacoum, with method detection limit (MDL) values dependent on sample type (Table 1). The Landcare Research brodifacoum analyses used HPLC with fluorescence detection, with methods developed for different sample types based on those described by Hunter (1983), Booth *et al*. (1999), and Primus *et al*. (2001).

RESULTS

No brodifacoum was detected in 217 water samples from Maungatautari, in any of the four water samples tested from Little Barrier, or in the four drinking water samples from Motutapu. On Little Barrier Island, bait pellets in exclusion cages were nearly completely disintegrated by 100 days after bait application. Soil samples from a grassland site on Little Barrier had residues of 0.2 ppm (n=2 with the same concentration) on day 56 and 0.03 ppm on day 153. Soil samples from a forested site had residues of 0.9 and 0.5 ppm on day 56 and 0.07 ppm on day 153. Brodifacoum was not detected in any of the paua and scallop samples from Little Barrier, or in pipi or mussel samples from Motutapu and Rangitoto.

On Little Barrier Island, track searches recovered carcasses of a blackbird (*Turdus merula*) and a pukeko (*Porphyrio melanotus*). Grid searches recovered carcasses of two blackbirds, four pukeko, 14 morepork (*Ninox novaeseelandiae*), one harrier (*Circus approximans*), two North Island brown kiwi (*Apteryx mantelli*) and two kakariki (*Cyanoramphus spp.*). The carcasses were too degraded for necropsy or liver sampling, except for one kiwi where necropsy gave a provisional diagnosis of bronchopneumonia with residual brodifacoum concentration in the liver of 0.26 ppm.

Following the Rangitoto /Motutapu eradication, no brodifacoum was detected in five dolphins or their stomach contents or in whole-body samples of pilchards collected from local beaches during July 2009. In some cases, degradation of penguin carcasses precluded necropsy. Of the seven penguins examined, there were no obvious signs of anticoagulant poisoning (such as haemorrhage) and in three of these necropsy indicated poor condition, i.e. no body fat, empty stomach. Of the total nine penguin livers tested, no brodifacoum was detected in six, but in three there were concentrations of 0.005, 0.007 and 0.17 ppm, respectively.

DISCUSSION

Brodifacoum in water

The water monitoring implemented at Maungatautari (217 samples tested, no brodifacoum detected) appears the most comprehensive reported to date. Brodifacoum was also not detected in water samples from Little Barrier and Motutapu, consistent with previous small-scale monitoring on Red Mercury Island (Morgan and Wright 1996) and Lady Alice Island (Ogilvie *et al.* 1997). Interacting factors likely to have contributed to such results are brodifacoum's overall low water-solubility (0.24 mg/l at 20ºC and pH 7.4, British Crop Protection Council 2000), adsorption of brodifacoum to organic particles (World Health Organisation 1995), and dilution with water volume and flow rate. If aerially applied baits were to enter fresh water, only a limited amount of the brodifacoum in them would enter solution, being more likely to remain bound to bait or to other organic particles present in the water or sediment. Binding of brodifacoum would render it undetectable in water that could have been used for drinking supplies.

Bait degradation and brodifacoum in soil

Bait degradation on Little Barrier took a similar time to that described by Craddock (2003a) at Tawharanui (NZ) where 96.5% pellets had completely broken down by 120 days in open grassed area, although bait degradation was slightly slower in a forested site. Thus a universal degradation time for all situations cannot be defined, especially as rainfall (Bowen *et al.* 1995), among other climatic factors affecting degradation, can vary from island to island. In each instance, monitoring should ensure that uneaten baits have degraded sufficiently to no longer present a non-target hazard. Following aerial bait (Talon 20P) application on Red Mercury Island (Morgan and Wright 1996) and Lady Alice Island (Ogilvie *et al.* 1997), no brodifacoum was detected in topsoil sampled at one month and over days 2 to 34, respectively. Those soil samples are presumed not to have been specifically associated with degrading bait, noting that brodifacoum is relatively immobile in soil (Eason and Wickstrom 2001). Hence, any residual soil concentrations are most likely to be localised around uneaten, degrading bait, as indicated by the Little Barrier results. The relatively low brodifacoum concentrations (<1 ppm) in these samples may have been due to the presence of disintegrated bait particles in the sample, in addition to limited movement of brodifacoum from bait into the soil. A decrease in the concentrations (from maximum 0.9 ppm to minimum 0.03 ppm over *C.* 100 days) suggests degradation in soil over time. Degradation rates of brodifacoum in a sandy clay loam was estimated as 22.4 weeks (US EPA 1998), but probably varies with soil type at least. Thus soil invertebrates near degrading bait on Little Barrier may have been exposed to low brodifacoum concentrations for a limited period. While exposure of laboratory earthworms (*Apporectodea caliginosa*) to 500 ppm brodifacoum in soil resulted in 85% mortality after 28 day's exposure (Booth and Fisher 2003), this soil brodifacoum concentration was 25 times higher than that of bait. It is unknown whether soil concentrations in a much lower (*c.* 1 ppm) range, more representative of field results, would affect soil invertebrate survival or health, and for how long sublethal residual concentrations of brodifacoum persist in soil invertebrates.

Brodifacoum in marine shellfish

Following accidental spillage in 2001 of 18 tonnes of PestOff 20R into the ocean at Kaikoura, NZ, brodifacoum residues were detectable for some weeks in marine shellfish commonly harvested for human consumption (Primus *et al*. 2005), which raised awareness and concerns about potential human exposure. An important point of difference was that the spill comprised an extremely large quantity of bait entering the ocean at one point. In contrast, aerial application disperses individual pellets, resulting in much smaller quantities of brodifacoum entering the ocean around island shorelines. The results reported here suggest that contamination of marine shellfish is unlikely following aerial application of brodifacoum baits for rodent eradication. That there were no detectable results in marine shellfish following the Little Barrier and Rangitoto/ Motutapu eradications is consistent with previous small monitoring efforts following bait applications on New Zealand islands. Two oyster samples and three of four mussel samples from Motuihe Island in 1998 were <MDL, with one mussel sample reported as 0.02 ppm as a conservative interpretation by the analysing laboratory (Landcare Research) against the detection limit available at the time. Two mussel samples from aquaculture farms near Great Barrier Island (Hauraki Gulf) were also below detectable concentrations, following a 2008 rat eradication attempt.

There is a lack of information regarding potential differences in exposure pathways between sediment and water-column-feeding shellfish species and the persistence of residual brodifacoum in shellfish. On this basis, residues may still be found in marine shellfish following aerial bait application, but the evidence so far suggests that the risk of secondary brodifacoum exposure to humans harvesting and eating shellfish is relatively low. Where this is a concern for proposed eradications, stipulating a no-harvest period linked to post-application monitoring is a prudent approach to confirming that there is no potential secondary human exposure as a result of consuming shellfish.

Brodifacoum in non-target wildlife

Brodifacoum is highly toxic to mammals and birds (Erickson and Urban 2004). Consequently, rodent bait presents a primary poisoning hazard to non-target mammals and birds. If exposure to the baits is not lethal, residual brodifacoum can persist for months in the livers of mammals (Eason *et al*. 2002; Fisher *et al*. 2003; Spurr *et al*. 2005) and birds (Fisher 2009), but is eliminated within days from blood and other tissues (e.g., Fisher 2009). Liver residues and stomach contents containing partially digested brodifacoum bait present the highest secondary hazard for mammalian and avian species that prey on rodents or scavenge carcasses (e.g., Howald *et al*. 1999; Shore *et al*. 1999). Some terrestrial invertebrates will feed on cereal-based bait and then contain residual concentrations of brodifacoum (e.g., Booth *et al*. 2001; Craddock 2003b; Bowie and Ross 2006). Secondary mortality of insectivorous New Zealand dotterels (*Charadrius obscurus aquilonius*) may have been through this environmental pathway (Dowding *et al*. 1999). Unpublished evidence of suspected secondary brodifacoum poisoning of two tuatara (*Sphenodon punctatus*) held in a zoo was the basis for implementing several mitigation measures to prevent brodifacoum exposure of tuatara held in outdoor enclosures on Little Barrier.

The 27 bird carcasses found on Little Barrier were of species previously reported as non-target mortalities in other New Zealand eradications using brodifacoum (e.g., Towns and Broome 2003), and in the absence of residue testing or necropsy data, the conservative assumption is they represent non-target mortality. Of 10 radio-tagged little spotted kiwi (*Apteryx owenii*), one was confirmed to have died of brodifacoum poisoning following rodent eradication on Kapiti Island, with haemorrhage found at necropsy, and with liver residues of 1.2 ppm (Robertson and Colbourne 2001). Wild kiwi have occasionally been recorded eating softened or degraded cereal bait, but their main prey are soil invertebrates such as earthworms, cicada nymphs and grass grubs (Robertson *et al.*1999), so primary and secondary exposure to brodifacoum was possible for the two brown kiwi found dead on Little Barrier Island. Better understanding of invertebrates as a residue vector is required to identify the most likely pathways of environmental exposure by kiwi to brodifacoum, and also to direct improved non-target risk mitigation measures for insectivores. Most morepork carcasses were found in areas where historical densities of Pacific rats had been highest, so presenting a possible increased risk of secondary poisoning. Since the bait application in 2004, morepork have remained abundant on Little Barrier and kiwi surveys show that the non-target mortality following the eradication did not have a population-level effect (Wade 2009). However, while this outcome supports an overall, long-term ecological benefit of rodent eradication to these populations, some community groups consider that any non-target bird mortality (especially iconic native species) is unacceptable.

The presence of residual brodifacoum in livers of three of nine penguins cannot be confirmed as sourced from the Rangitoto/Motutapu bait applications. Brodifacoum bait stations are commonly used for commensal rodent control in New Zealand, and also for field use against brushtail possums and rodents (see Hoare and Hare 2006). Exposure of the penguins to brodifacoum before the Rangitoto/Motutapu aerial operation cannot be ruled out because brodifacoum was almost certainly being used in the Hauraki Gulf area, potentially around buildings or on boats in coastal areas near terrestrial penguin habitat, before June 2009. The presence of brodifacoum in the penguins also cannot be confirmed as a direct cause or contributor to their mortality, as brodifacoum can be retained in liver at sublethal concentrations, as reported in a range of live-sampled, apparently healthy mammals and birds (see Fisher 2009). Relatively high liver concentrations (< 1 ppm) are more strongly associated with lethal exposure, but there is overlap between the lowest lethal and highest sublethal concentrations reported. For example, Littin *et al.* (2002) measured concentrations as low as 0.33 ppm in livers of lethally poisoned possums, but sublethally exposed chickens (*Gallus gallus*) had liver residues of 0.45-1.00 ppm (Fisher 2009). Rather than estimating a threshold liver concentration definitive of lethal brodifacoum exposure (e.g., Kaukeinen *et al.* 2000), it is more valid to attribute increasing certainty of lethal exposure with increasing liver concentration. For example, Myllymäki *et al.* (1999) estimated that survival probability in voles (*Microtus* sp.) started decreasing at 0.20 ppm in liver. Necropsy observations of fresh carcasses may assist in determining the cause of death (e.g., Hosea 2000; Stone and Okoniewski 2003), and in some cases can be supported by information on the circumstances of carcass recovery and expert knowledge of common causes of mortality in the species concerned.

The 0.26 ppm liver concentration in the kiwi from Little Barrier Island was in the ‘overlap’ concentration range with low certainty, but possible lethal exposure. While necropsy did not indicate haemorrhage, the recovery of the carcass in the operational area soon after bait application and previous confirmation of kiwi mortality in similar circumstances (Robertson and Colbourne 2001) support a conservative diagnosis of brodifacoum poisoning. In all of nine penguin carcasses found on beaches outside the operational area in the month following the Rangitoto/Motutapu operation, necropsy indicated starvation with no evidence of haemorrhage considered typical of anticoagulant poisoning. In some years, many little blue penguin carcasses are washed ashore in New Zealand, probably as the result of food shortage or biotoxins (e.g., Heather and Robertson 1996). For the six penguins in which no brodifacoum was detected, starvation was the most likely cause of death. In two of the three penguins with detectable liver residues, starvation was also most likely because the very low brodifacoum concentrations of 0.005 and 0.007 ppm were most representative of sublethal exposure. The penguin with 0.17 ppm liver concentration was within the ‘overlap’ range with low-certainty, but possibly lethal exposure. Because the carcass was found outside the operational area and with no haemorrhage seen at necropsy, the known seasonal starvation in local penguin populations was considered the more likely cause of death than brodifacoum poisoning. However, it is unknown whether brodifacoum exposure in this penguin was a contributing factor to mortality.

Importance of monitoring

While environmental sampling and subsequent analysis adds labour and operating cost to eradication programmes, monitoring data from completed eradications have undoubted value in supporting future risk assessments. When budgeting to cover mandated monitoring, generally as stipulated by the conditions of a regulatory approval, eradication planners should retain the flexibility to obtain additional environmental samples that can be stored pending analysis; it is better to have samples that don’t need testing than to need to test and not have samples. Even if the potential for brodifacoum contamination is considered low, directly addressing concerns through analysis for residues may have greater ‘public relations’ value than the dollar cost of a laboratory test, especially if confirmation or assurance is provided by nil-detected results from a locally relevant environment. Where brodifacoum is detected in environmental samples, this contributes to future risk assessments and mitigation approaches. The detection of residual brodifacoum in little blue penguins shows the role of monitoring in identifying new information. In this case, it has raised questions about the pathways and extent of exposure in penguins and the significance of sublethal residual concentrations for longer-term survival fitness. The Rangitoto/Motutapu bait application also attracted media attention and public concern that contributed to increased publicising of both factual and inaccurate information about brodifacoum and its effects. For managers planning eradications on inhabited islands, failure to clearly address the information gaps identified by community concerns around the aerial application of brodifacoum will mean that clear justification of eradication benefits will become increasingly difficult.

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Wetapunga (*Deinacrida heteracantha*) population changes following Pacific rat (*Rattus exulans*) eradication on Little Barrier Island

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Abstract Wetapunga (*Deinacrida heteracantha*) (Orthoptera: Anostostomatidae) is the largest of the 11 giant weta species found in New Zealand and is listed as Nationally Endangered. Originally distributed throughout Northland and Auckland, it is now restricted to Hauturu (Little Barrier Island; 3083ha). Largely arboreal, wetapunga feed mostly on the foliage of a range of plants by night and hide in refuges during the day. Following the eradication of kiore or Pacific rat (*Rattus exulans*) the abundance of wetapunga was recorded at fixed sites over five years. During May each year, 121 person hours were spent searching the same areas for the same length of time by the same three-person team to provide consistent search effort. Search time was split approximately 50:50 between night and day. Wetapunga encounters more than doubled over the five years with approximately 50% increase every second year. The size classes of wetapunga found were biased towards adults or near adult instars. On average, approximately 25% of all adults found each year were in male-female pairs, either pre-, or post-mating, or actually in copulation which appears to last for about 24 hours. Some adult wetapunga were found during the day in fully exposed positions indicating their behaviour may have changed due to reduced predator pressure following Pacific rat eradication. The increasing wetapunga numbers over the study period reflect the benefit of rodent eradication and are consistent with other studies on the impacts of exotic rodents on New Zealand indigenous large bodied, flightless, nocturnal invertebrates.

Keywords: Flightless invertebrates, Hauturu, monitoring; surveying, population recovery

INTRODUCTION

The Little Barrier Island giant weta or wetapunga (*Deinacrida heteracantha*) (Orthoptera: Anostostomatidae) is New Zealand's largest weta species (Gibbs 1999) and is slow moving, flightless, nocturnal and largely arboreal in forest. Early biologists reported the species as widely distributed throughout Northland, Auckland, and on Great Barrier Island (Colenso 1882; Dieffenbach 1843; Buller 1895; Hutton 1897). However, the species is now restricted to the 3083 ha, forest covered Hauturu (Little Barrier Island) Nature Reserve. Wetapunga is a species of high conservation value and is listed as Nationally Endangered (Hitchmough *et al*. 2007).

Surveys on Hauturu located wetapunga at night on the foliage of tree species (Richards 1973; Meads 1990; Meads and Balance 1990; Meads and Notman 1993; Gibbs and McIntyre 1997; Gibbs 2001), but rarely found the weta during daytime searches of large cavities that could be used as refuges. Richards (1973) and Meads and Notman (1993) considered it easiest to locate trees containing wetapunga by examining the ground beneath them for faecal pellets. Despite finding these characteristic, very large pellets, relatively few wetapunga were ever seen although considerable time was spent searching (Meads and Notman 1993; Gibbs and McIntrye 1997).

Following several intensive surveys on Hauturu, Gibbs and McIntyre (1997) considered wetapunga poor candidates for the use of artificial refuges (Trewick and Morgan-Richards 2000) to estimate density. Some years after the current study was initiated a novel technique for detecting wetapunga involving the use of tracking tunnels was reported (Watts *et al*. 2008). While this technique is a breakthrough in detecting the presence of giant weta, its ability to monitor population density has yet to be proven.

At the beginning of our study there was thus no accepted standard monitoring technique for wetapunga other than to employ experienced searchers for labour intensive field observations (Gibbs and McIntyre 1997).

Cats (*Felis catus*) were introduced to Hauturu around 1870 but were eradicated by 1980 (Veitch 2001). Kiore or Pacific rat (*Rattus exulans*) is known to have negative impacts on a range of invertebrate species (Green 2002; Towns 2009). Wetapunga surveys during the 1990s appeared to show a decline in abundance, which led to concern that the combined effects of Pacific rats and a recent reintroduction of the insectivorous North Island saddleback (*Philesturnus carunculatus*) in 1984 (Lovegrove 1996) may have been involved (Gibbs and McIntyre 1997). Pacific rats were eradicated from Hauturu in 2004 (Bellingham *et al*. 2010). Since wetapunga is a Nationally Endangered species, its' response was included as a measure of the benefits or outcomes of the eradication. Here we describe changes in wetapunga populations during the first five years following Pacific rat eradication.

METHODS

Annual surveys of wetapunga on Hauturu were carried out from 2005 to 2009 for one week each May, which is when Gibbs and McIntrye (1997) found the largest number of individuals. Search areas comprised 10 forest locations of variable size (all < 1 ha), mostly within 1 km of the ranger's residence/base on the island. All sites were in regenerating kanuka - broadleaf forest at the base of the tracks and stream valleys indentified in Fig. 1. All forest sites were considered suitable wetapunga habitat. During each survey, a total of 121 person hours were spent searching the 10 sites for the same length of time by the same three-person team to provide a consistent search effort. The same site received the same search effort each year, with more time usually allocated to the larger sites. Search time was split approximately 50:50 between day and night, with the former carried out after sunrise and the latter during the first six hours of darkness.

All wetapunga were located visually without the use of traps or lures. Day searching concentrated on any likely above-ground refuge sites such as in the dead fronds of nikau palm (*Rhopalostylis sapida*); at the base of live fronds and in dead fronds of the treefern species silverfern (*Cyathea dealbata*) and mamaku (*Cyathea medullaris*); in cavities under bark and in thick dead brush of kanuka

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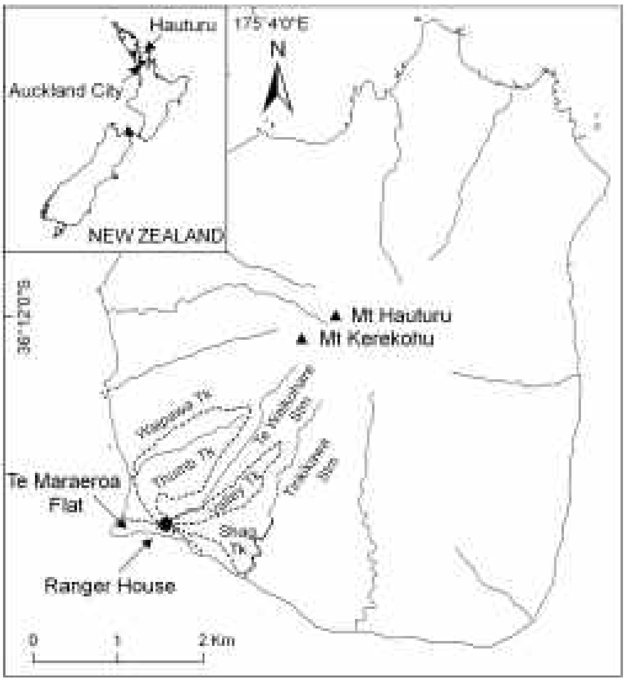

Fig. 1 Hauturu (Little Barrier Island). Wetapunga surveying occurred in 10 forest areas predominantly within 1 km of the Ranger House.

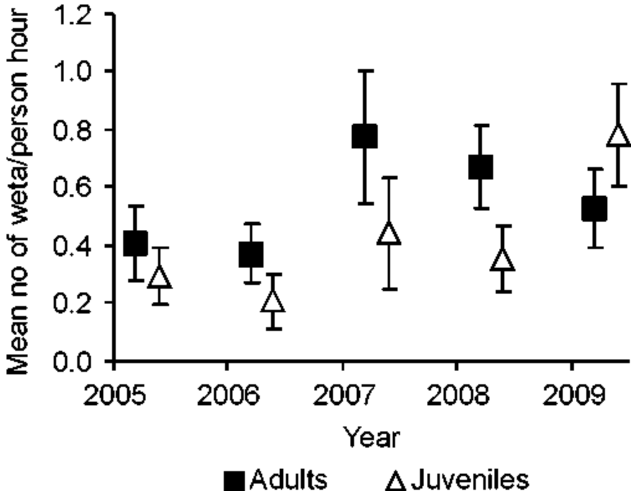

Fig. 2 Mean number of weta per person hour found each year across all sites over all search times. Error bars are standard errors.

(*Kunzea ericoides*); within dead hanging foliage of Collospermum (*Collospermum hastatum*); in any hollow trees or branches; or in thickets of dense foliage. Night searches used headlamp light beams to locate wetapunga on foliage, trunks and branches, as well as on the ground.

All wetapunga were collected and the right hind femur length measured to determine the instar. The sex was noted as well as the proximity of other wetapunga, particularly any male-female pairs. To avoid repeat recordings, each weta was marked with a Xylene-free marker pen. Weta were released in the exact location where found with particular care being taken to ensure juveniles were well hidden after release. Searching was discontinued during periods of persistent rainfall.

Data were analysed in the statistical programme 'R', version 2.9.2 (R Project, www.r-project.org), checked for normality, and are presented with standard errors. Weta counts were grouped as either adult or juvenile wetapunga. These were analysed using a linear mixed effect model with the number of weta as the response variable; year, time of day (day/night) interaction as the explanatory variable; and site as the random effect.

RESULTS

Wetapunga were found to be widely distributed on host plants and in refuges within the forest during day and night searches. Following the Pacific rat eradication in 2004, the total number of wetapunga found more than doubled from 78 in 2005 to 171 in 2009. Very low numbers of early instars (one to six) during all but the final year prevented meaningful analysis of temporal trends in each instar. Therefore the data for all juvenile instars were pooled for analysis. There was a significant increase in juvenile wetapunga over the five years (T_{64} = 2.99, P = 0.004) while the increase in adults was less pronounced (T_{64} = 2.12, P = 0.03) (Fig. 2). The mean number of adult weta doubled between 2006 and 2007 but then did not change substantially in each of the following three years (Fig. 2). Total numbers increased by approximately 50% every second year with the majority of weta being adult or late instar (eight or ninth) juveniles. As expected with such a very large but cryptic invertebrate, there was a positive relationship between weta age and numbers found, since larger weta were the most likely to be found. In the instars old enough to determine sex (fifth instar or older) there was a consistent 50:50 sex ratio recorded each year.

Over the five year study period, on average approximately 25% of all adults were found as male – female pairs, either copulating or beside each other, indicating likely pre-mating or post-mating behaviour. During the day, many copulating pairs were found in sites with little or no cover. Some were found fully exposed on large kanuka tree trunks and this behaviour was seen consistently each year. Sometimes a second male was found within 2-3 m of the pair. Copulation commenced in the evening and appeared to last for approximately 24 hours.

Over the five year period there was no significant difference in numbers of adult or juvenile weta found during the day versus night searches (T_{64} = 1.47, P >0.05 and T_{64} = -0.72, P > 0.05 respectively).

Despite searches in a wide variety of potential daytime refuge sites, weta were usually located in well protected refuges. Generally, refuge sites were at least 1 m above ground with only a few weta found lower during the entire survey period. Preferred sites appeared to be associated with foliage with colour and patterns that afforded wetapunga of all sizes extremely good camouflage. This was particularly apparent in dead, hanging silver fern fronds where weta were very difficult to detect and could only be found by silhouetting the frond against the sky to detect the weta shape. Where these dead fronds were in the form of a joined 'skirt' the individual fronds needed to be teased apart to find wetapunga within. Despite careful examination of these fronds, only adult or late instar juvenile weta were found within them during the day.

At night, wetapunga of all ages were seen out on foliage, on branches and trunks leading to foliage, which suggests that weta move from arboreal refuge sites to foliage on which to feed. Despite their large size, adult wetapunga moved nimbly along surprisingly thin twigs and branches. Wetapunga were found on live foliage of karamu (*Coprosma robusta*), mamangi or tree coprosma (*C. arborea*), mahoe (*Melicytus ramiflorus*), hangehange (*Geniostoma rupestre*), kohekohe (*Dysoxylum spectabile*), rimu (*Dacrydium cupressinum*), kawakawa (*Macropiper excelsum*) as well as nikau and silverfern. We have not determined whether the frequency of weta sightings varied by tree species.

DISCUSSION

Our encounter-based search method revealed significant and consistent increases in the abundance of wetapunga on Hauturu following the eradication of Pacific rats. Other methods of detecting giant weta include the use of tracking tunnels (Watts et al. 2008), but these only indicate the presence of weta and as yet cannot provide robust population density measurements.

Previous surveys for wetapunga on Hauturu concentrated on night searching (Meads and Balance 1990; Meads and Notman 1993; Gibbs and McIntyre 1997), often using faecal pellets on the ground to indicate likely weta presence in foliage above (Richards 1973; Meads and Notman 1993. Pilot surveys by CJG (unpublished) indicated that wetapunga could be found in daytime refuges. We also found that faecal pellets were an unreliable indicator of weta activity, because only the freshest pellets indicated nearby individuals. Wetapunga were found regardless of whether we found pellets. This is probably because the arboreal habits of wetapunga can lead to pellets landing away from their source, as well as the wetas' mobility which can take them far away from the point of defecation.

Unlike earlier researchers, we found wetapunga at the rate of up to one per person hour search time during the day from the first year onwards. The search team probably became more proficient at locating wetapunga during the day as the first survey proceeded, but few additional daytime refuge site types were located in subsequent years. Furthermore, plant species such as tree ferns and nikau palms were consistently searched each year. Any improvements in search proficiency are unlikely to account for the more than doubling of the numbers of wetapunga recorded over the five year study.

The total number of wetapunga increased by approximately 50% every second year. Except for 2006 and 2007, this increase was largely driven by increased numbers of juveniles (Fig. 2). Why have adults not shown the same increased abundance as for juveniles? We suggest that since the rodent eradication, adult wetapunga have become more mobile in response to decreased predation pressure. Human visual range for large instars of wetapunga in these forests, which have a canopy height of about 15 m, is restricted to about 2 m during the day and perhaps double that at night. Other lines of evidence suggest that wetapunga are now using larger areas in the subcanopy and canopy where they cannot to be found by our search methods.

Radio-tracking studies suggest that wetapunga behaviour significantly changes after the final moult, when some adults travel over 50 m per night, apparently along the ground but also potentially over aerial walkways (Watts and Thornburrow 2009). In contrast, an earlier study by Gibbs and McIntyre (1997) with transmitters fitted to a few sub-adult male and female weta revealed sedentary behaviour, with just short movements to and from feeding sites close to refuges. Our study had repeated observations of several marked individuals, which confirmed the sedentary nature of large nymphs. The more recent radio-telemetry work also showed that 83% of the daytime refuge sites for adults were greater than 2 m off the ground (Watts and Thornburrow 2009). These studies indicate that, compared with sub-adult or younger instars, adult weta are substantially more mobile, make more extensive use of the entire forest structure, and are likely to be more difficult to observe from the ground. Thus we believe that the relatively low level of increase in adults compared to juveniles over the five year study period could be a reflection of relaxed predator pressure and increased adult vagility. Regardless of the mechanism, many more adult wetapunga are now being seen than were found during previous surveys while rodents were still present.

In the present study, most juvenile weta were found in (day) or near (night) refuge sites associated with dead foliage of plants such as tree fern and nikau palm. Within the forest structure on Hauturu, most dead foliage of such plants was within 3 – 4 metres of the ground. Since much of this habitat was available to us for searching, and if favoured by juvenile wetapunga then we probably had access to a greater proportion of juveniles than adults. Therefore the increased numbers of juveniles that we observed may provide a more accurate indication of wetapunga population trends.

The many adult wetapunga that we found as pairs is likely related to the early winter season of the surveys and approximates the 28% of weta radio-tracked as pairs by Watts and Thornburrow (2009). Many of the pairs in both studies were found either fully exposed or with relatively little cover to protect them from potential predators, including some pairs on the trunks of large kanuka trees in full view 1-3 m above ground. With copulation likely to last at least 24 hours, such behaviour in the presence of rats likely made these weta extremely vulnerable to predation. We also occasionally found individual adult wetapunga in relatively open positions with little or no cover, whereas surveys during the 1990s in the presence of Pacific rats made no such observations. Similar changes in conspicuousness have been recorded for several other weta species following rat eradications (Bremner *et al*. 1989; Rufaut and Gibbs 2003). Such observations indicate that the behavioural and morphological defences weta have against most natural bird predators are less effective against introduced mammals.

Invertebrates caught in pitfall traps immediately following the eradication of Pacific rats on Tiritiri Matangi Island (Green 2002) showed increased numbers of a range of nocturnal, flightless, large bodied species, including the ground weta *Hemiandrus pallitarsus* (Orthoptera: Anostostomatidae). Captures of this species increased four-fold in the first six years following rat removal (Green unpubl. data). By comparison, the doubling of wetapunga numbers in five years seems conservative, although the population is still increasing. Further monitoring is required to determine the upper limit of wetapunga population growth.

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Monitoring of a population of Galápagos land iguanas (*Conolophus subcristatus*) during a rat eradication using brodifacoum

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Abstract Little is known about the toxic effects on reptiles of the anticoagulant rodenticide brodifacoum, which is often used to eradicate introduced rodents from islands. While many islands have both introduced rats and native lizard fauna, the impacts of large scale use of brodifacoum bait on native reptile populations have been largely unstudied. The population of Galápagos land iguanas (*Conolophus subcristatus*) on Seymour Norte, Galápagos, was monitored during a ship rat (*Rattus rattus*) eradication operation. Klerat bait (50 ppm brodifacoum) was applied at approximately 3 kg/ha in two applications in October 2007 in an apparently successful eradication. Six iguanas were found dead afterwards, apparently due to consumption of bait and/or poisoned rats, with overall mortality rate estimated at approximately 4.5% of the population. Iguana deaths were recorded more than two months after bait application, suggesting that reptile mortality may be delayed after a brodifacoum baiting operation and monitoring intervals need to be extended to detect this. The relatively low rate of poison application suggests rat eradication on arid islands may be achievable using less brodifacoum than elsewhere.

Keywords: *Rattus rattus*, density, arid, Seymour Norte.

INTRODUCTION

Introduced rats (*Rattus* spp.) threaten island faunas worldwide. In response, there have been numerous campaigns to eradicate these species in order to safeguard island populations of native birds, reptiles and invertebrates (Howald *et al*. 2007). The most efficacious method for eradication has been the application of 2nd generation anticoagulant rodenticides, particularly brodifacoum, either through aerial or hand broadcast of bait (Howald *et al*. 2007). In general there is a positive response in numbers and condition of native species after rat eradications (Parrish 2005; Daltry 2006; Towns *et al*. 2007; Olivera *et al*. 2010). However, applications of brodifacoum bait can put at risk some of the native species conservation managers are trying to save.

Brodifacoum is known to have non-target impacts on species of mammals and birds, but there is little information about its potential effects on reptiles at a population level through primary and secondary poisoning (Eason and Spurr 1995; Hoare and Hare 2006). Reptiles are known to consume cereal based rodent baits, which appear to be more palatable when wet (Merton 1987; Freeman *et al*. 1996; Marshall and Jewell 2007). On Round Island, Mauritius, during a 1986 rabbit (*Oryctolagus cuniculus*) eradication using cereal-based Talon 20P (20 ppm brodifacoum), at least 100 Telfair's skinks (*Leiolopisma telfairii*) died (Merton 1987). Only the largest lizards were apparently affected and deaths were recorded between three and six weeks after the poison was laid. However, there was no evidence of any effect at a population level and two years later Telfair's skinks were still numerous on the island. In contrast, although Wright's skinks *Mabuya wrightii* consumed Talon 50WB (50 ppm brodifacoum) and Talon 20P during a rat eradication in the Seychelles, no dead skinks were found despite searches for them (Thorsen *et al*. 2000). Secondary exposure of reptiles to brodifacoum was reported by Burbridge (2004), who noted that bungarras (*Varanus gouldii*) ate dead and dying ship rats (*Rattus rattus*) after a 1996 eradication on the Montebello Islands, West Australia, using Pestoff 20R pellet baits (20 ppm brodifacoum). Bungarras were apparently common and in some cases ate so many rats, their droppings were dyed green, presumably from the bait still present in the rats' gastrointestinal tracts. No dead or moribund bungarras were found despite active searches for dead animals. Similarly, there was no detectable decline in a monitored Selvagem Grande, Portuguese Madiera, population of endemic geckos (*Tarentola bishoffii*) after an eradication of mice (*Mus musculus*) and rabbits in 2002 (Olivera *et al*. 2010). The operation used hand-laid Pestoff 20R initially and later Talon wax blocks or Klerat wax block (50 ppm brodifacoum) in bait stations at an overall application rate of approximately 20kg/ha.

This limited evidence suggests that reptiles have a low risk of population-level declines through brodifacoum-induced mortality after rodent eradications. However, to our knowledge, there have been no direct measures of population density of reptiles immediately before, during, and after a field application of brodifacoum baits. Conceivably, populations could decline soon after an eradication through primary or secondary poisoning, or later through multiple year effects on survival or reproduction with potential adverse effects on population genetics. Galápagos land iguanas (*Conolophus subcristatus*: Iguanidiae) are a large (mean adult size: 100 cm) reptile that has undergone severe declines in abundance and distribution through predation by cats (*Felis catus*) and dogs (*Canis lupis familiaris*) and habitat destruction by goats (*Capra hircus*). On Seymour Norte, land iguanas were the only reptile capable of swallowing entire cubes of rodent bait to be used in an eradication attempt. The iguanas were also ideal for monitoring, because of their large size and terrestrial habits.

As a rat eradication was planned for Seymour Norte, we aimed to monitor the effect of brodifacoum on a large reptile in a more systematic manner than previous described and present the results to assist other pest eradications where anticoagulants use was planned and native reptile fauna may be at risk. In this paper, we describe the potential effects of the exposure of iguanas to brodifacoum during an eradication campaign against rats. We undertook small trials with captive iguanas presented with rodent bait and poisoned rodent carcasses. We also conducted a larger field study to investigate potential immediate effects of brodifacoum exposure on iguanas and measure the species' abundance during the rat eradication and over the subsequent six months. We also aimed to determine whether baiting had detectable delayed effects by searching for dead or moribund individuals after the operation.

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METHODS

Poison trials with captive land iguanas

Prior to carrying a rat eradication on Seymour Norte, small-scale bait acceptance trials were carried out on land iguanas in captivity in Charles Darwin Foundation enclosures at Puerto Ayora, Santa Cruz, Galápagos Province, Ecuador. Two young land iguanas deprived of other food for two weeks, were only offered shredded Klerat and a single adult male land iguana was deprived of food for three weeks, at the end of which it was offered five cubes of Klerat. After a further two weeks of fasting the male iguana was then offered a fresh corpse of a ship rat poisoned with Klerat.

Field eradication site

Isla Seymour Norte (184 ha, 90° 17' W, 0° 23' S) north of Baltra and Santa Cruz Islands, Galápagos Province, Ecuador (Fig. 1) is a raised basaltic platform overlaid with a thin layer of soil and open forest of *Opuntia echios* cactus, *Bursera malacophylla*, *Parkinsonia aculeate* and *Scalesia crockeri* (Hamann 1979). Rainfall is highly variable (mean annual precipitation of 228 mm) and mainly in the 'hot' season from January to June.

Native fauna includes Galápagos land iguanas, Galápagos lava lizards (*Microlophus albemarlensis*), and sea bird species such as blue-footed boobies (*Sula nebouxii*) and great frigatebirds (*Fregata minor*). Ship rats were known to be present since 1986 and probably invaded from the nearby Baltra Island, where ship rats and mice are present.

The proposed eradication of ship rats used a hand broadcast of Klerat a wax-based bait, coloured dark blue, with a loading of 50ppm of brodifacoum. A single cube of Klerat weighs 3.5-5 gm. Captive ship rats eat Klerat when offered with other natural food and Klerat would also be taken by free-roaming rats on the Seymour Norte (unpubl. data). Land iguanas are opportunistic omnivores, often feeding on carrion in addition to their normal diet of *Opuntia* vegetation and fruit, and are at a high risk of poisoning through eating Klerat or poison-killed rats.

Fig. 1 The Galápagos Islands showing the location of Seymour Norte.

Bait take and rat carcass removal by iguanas

From 5-12 September, a small-scale poisoning trial was conducted on the island to investigate bait take by iguanas. Bait was hand laid across 2.5 ha in one morning in piles of 10-15 cubes every 20 metres following lines 20 metres apart to simulate conditions of the planned eradication. Over the next eight days and in subsequent visits all iguana droppings in the area were checked for signs of the blue bait. Ten rats caught on the trapping grid were used to investigate consumption of rats by iguanas. The bodies were placed in the vicinity of male iguanas' digging burrows to observe removal of rats.

On 7 November 2007, after the second bait application, six people on foot searched Seymour Norte for dead or moribund iguanas and/or fresh iguana droppings to detect any bait consumption along east-west transects 100 m apart. If any blue droppings or dead iguanas were found, surveyors in that transect stopped for 5-10 minutes and thoroughly inspected the surrounding area looking for more blue droppings or dead iguanas. A post-monitoring trip on 2-5 January 2008 counted iguanas using the transect lines established in September 2007.

Rat density

From 5-12 September 2007, a 10 x 10 grid of rat traps (Tomahawk live traps 40 x 12 x 12 cm) set at 25m intervals was established at the eastern end of the island. Traps were baited with a mixture of rolled oats and peanut butter, rolled into a ball within a small piece of grease proof paper. Bait was suspended from the top of the trap at the back with a short piece of wire to reduce interference by ants. Any rats trapped were humanely killed. To estimate the Effective Trapping Area (ETA) for the rats, a boundary strip was added to the edge of the trapping grids (Dice 1938). The width of the boundary strip was set by adding the radius (56 m) of a circular average home range of ship rats from a forested habitat (Hooker and Innes 1995). An approximate density was estimated by dividing the total number of rats caught by the ETA. This calculation assumed that during an intensive, short period of trapping immigration, emigration and reproduction by rats would be nil (Brown *et al*. 1996).

Monitoring land iguana population

In order to detect changes in the abundance of land iguanas on Seymour Norte, estimates of population density were made before and after the eradication. Thirty 200 m transects were marked out across the island. On 20 September 2007 all iguanas, and the distance each one was from the transect line, was recorded. The transects were sampled again on 2-5 January 2008. Using these data the density of iguanas was calculated using the Program DISTANCE (Buckland *et al*. 1996; Thomas *et al*. 2006).

Eradication operation

Klerat was first applied on 10th October 2007. Pre-programmed points at 25 m intervals along lines 25 m apart were loaded into personal GPS units for each person broadcasting the poison and 15 cubes of Klerat deposited at approximately 25m intervals along east-west lines across the entire island. Bait was deposited close to low vegetation and other cover, rather than leaving it on open soil. Baits were also deposited around the coast. Lines logged by each person were downloaded onto a computer at completion of the transect lines and the map checked for gaps in coverage. Any gaps were then located and extra poison bait laid in the gaps. The second application of poison bait was on 30 October. Because the first application of poison required less time than expected, an additional east-west line of bait was broadcast between the earlier bait lines. Once the east-west lines were completed, a north-south set of lines was used with baits distributed in the same manner, but at a lower application rate than on 10 October. Two groups also applied bait along the high tide line and on coastal cliffs. After completion, any gaps in poison application were located and filled as on 10 October. On 7 November 2007, 46 poison bait stations were established on Mosquera Island, 475m south of Seymour, to create a barrier for rats invading from Baltra Island, which is 340m further to the south.

On the first poison application, approximately 250 kg of Klerat was applied at a rate of 1.40 kg/ha or 310 cubes/ha. On the second application, 20 days later, approximately 280 kg of Klerat was applied at a rate of 1.52 kg/ha. A total of 530 kg/ha of poison bait was broadcast on Seymour Norte at a rate of 2.92 kg/ha

RESULTS

Poison trials with captive land iguanas

Neither of the two young iguanas accepted the Klerat and the adult male did not consume the five Klerat cubes. However, this animal did consume the poisoned rat with no apparent ill effects on behaviour or activity.

Bait take and rat carcass removal by iguanas

No fragments of bait or blue-coloured droppings were found immediately after the initial small-scale poisoning trial or in subsequent visits. Of the ten dead rats placed next to male iguana burrows: three were removed over the next six days from one site by a short-eared owl (*Asio flammeus*); another disappeared by the next visit two weeks later; and none of the remaining rats were moved from their original locations.

Before the second bait application, blue-coloured iguana droppings were found on Seymour Norte, which indicated that some iguanas had eaten bait or dead rats that contained bait. The latter possibility seemed unlikely as no hair or bone was found in these droppings. Some droppings contained Klerat in cubes, which suggests ingestion of baits with little chewing by the iguanas. Of 91 recent iguana scats, five in one group on the coast contained Klerat, suggesting that one animal was responsible. With so few droppings with Klerat, the decision was made to continue with the operation. No Klerat was found on the island by January 2008, despite several days searching.

Rat density

Over six days of trapping, 49 rats were caught before captures declined to zero. The ETA of the rat trapping grid was calculated as 14.01 ha. Assuming that we caught all rats within the ETA over six days, the approximate density of rats on the grid was 3.5 rats/ha (95% C.I.: 2.8-6.4 rats/ha). Of the captured rats, 27 were males (4 juveniles) and 22 were females (4 juveniles). No rats were detected subsequent to the eradication operation on eight lines of 25 live-traps for 3 nights (600 TN). The most recent negative result was 18-21 March 2009.

Iguana population estimates

The DENSITY programme suggested that a Uniform Cosine model provided the best fit for estimating iguana density on Seymour Norte and indicated a detection probability close to 1.0 out to the truncation point at 10 m from the transect lines. The pre-poison population estimate was 2467 (95% C.I.: 1744-3397) and the post-operation estimate 2222 (95% C.I.: 1816-2718), indicating a potential population decline of up to 9.9%. However, because the CIs of the second estimate fall within the range of the first, there is no statistical support for any difference in the population estimates.

During the post-operation monitoring on 7 November, 263 live and no dead iguanas were detected on the east-west transect. Two dead rats were located.

The first post-eradication monitoring trip on 2-5 January 2008 located six dead and 128 live iguanas on the 200m transects. Two iguanas were desiccated, and four had died more recently, two of which were located down burrows due to the smell. Two of the carcasses had blue paste or whole cubes in the alimentary tract but no bones or fur, which suggested consumption of Klerat only. These results indicated an observed mortality rate of 4.7%.

DISCUSSION

There was some loss of Galápagos land iguanas from bait ingested after the hand broadcast of brodifacoum on Seymour Norte. If based on population estimates, there may have been a decline in density of up to 9.9%. This is the worse-case projection and lacks statistical support. A more supportable estimate derived from the population census, where the observed mortality was 4.5%. The pre-eradication trials suggest that iguanas were not likely to eat the bait and the presence of only a few iguana droppings in discrete piles, suggest that only few individuals were eating the bait or dead rats. Blue objects, like Klerat, are not a preferred colour for some reptiles (Tershey and Breese 1994) so this may also explain why only a few blue scats were discovered. The lack of bait take by captive animals may be due to the very small sample size or better body condition, whereas in a larger population more diverse foraging behaviour or interspecific competition may predispose island iguanas to more opportunistic prey sampling.

Two months after the eradication, four recently dead iguanas were found. It is unknown whether death was caused by ingestion of brodifacoum because no samples were taken for analysis. If poison was responsible, it may have taken at least six weeks to kill the iguanas, unless iguanas found bait two months after the operation. Delayed mortality was found for Telfair's skinks on Round Island, Mauritius, three to six week after a poison bait application, often during particularly hot days or times of day (Merton 1987). The possibility that some reptiles have delayed effects from brodifacoum due to some aspect of their physiology that differs from birds and mammals (Merton 1987) deserves further research. Monitoring of poison effects on reptiles susceptible to bait intake should thus be extended to detect possible delayed mortality several months after application. This would reveal situations where reptiles die of chronic toxic poisoning during the post-operation period, rather than from immediate acute poisoning commonly documented in mammals and birds. We were unable to find any information on the effects of brodifacoum on snakes, geckos and many smaller lizards. Because our limited data suggests primary poisoning was the principal reason of death for iguanas, trials also need to be undertaken at higher bait application rates where an increased encounter rate may mean more bait is ingested by reptiles. Moreover, consideration should be given to sampling subdominant animals in less optimal habitat that may be more likely to eat poison bait than well-fed animals in prime habitat. Trials also need to investigate possible sub-lethal effects of brodifacoum exposure on reproduction and foraging in reptiles which may have long-term effects not shown by this research. The sparse available results suggest that the effects of brodifacoum on reptile populations are limited. However, until more research like radio-tracking or mark–recapture studies is conducted, conservation managers should consider non-target risk mitigation measures specifically for herbivorous or carrion feeding reptiles when using brodifacoum to eradicate rodents on islands with native reptiles.

It appears that eradications of rats on arid islands may be able to use quite low application rates of poison. Less than 3 kg/ha of Klerat was applied to Seymour Norte and after 18 months and one and a half breeding seasons (Clark 1980) rats were not present. The relatively low density of the non-breeding rat population on Seymour Norte and apparent palatability of the poison bait suggests that the population may be strongly food limited in the dry season (Clark 1980). Ship rat density in the Galápagos has a positive correlation with vegetation biomass (Clark 1980), so on islands with open, arid zone vegetation rat density should only be high during wet El Niño years when vegetation growth is substantial. In dry years eradications may be successful with low applications of bait which likely reduces non-target risk in addition to resources and time. Although Klerat has a higher loading of brodifacoum (50 ppm) than other bait formulations used for rat eradication operations (20-25 ppm) it was applied at a low rate (< 3 kg/ha) compared to previous eradications that used aerial application rates of 12 kg/ha or more (Empson and Miskelly 1999; McClelland 2002). This will substantially reduce the amount of resources and time required, as well as risks to non target species, and should be tested on smaller arid islands with a view to scaling up to larger arid islands (Cayot 1996; Harper and Carrion 2011).

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Restoration through eradication: protecting Chesapeake Bay marshlands from invasive nutria (*Myocastor coypus*)

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Abstract Coastal marshes on Delmarva Peninsula, Chesapeake Bay, Maryland, USA, provide valuable ecosystem services including flood prevention, erosion control, filtration, and carbon sequestration, and support commercial and recreational fishing, trapping, hunting, and ecotourism that generate billions of dollars for the region. Nutria (*Myocastor coypus*) were introduced to Dorchester County on the eastern side of Delmarva peninsula in 1943. They spread rapidly and reached peak densities in the late 1990s when vegetation studies linked nutria herbivory to massive wetland loss throughout Maryland's lower eastern shore. A coalition of state, federal and non-governmental organisations obtained congressional funding to eradicate nutria from the Delmarva Peninsula and, beginning in 2002, implemented a systematic eradication plan. The eradication team used integrated methods to complete the initial reduction of nutria populations on 60,000 ha of marsh in five counties across Maryland's lower eastern shore. Population reductions to near-zero were accomplished using trapping and shooting applied systematically using GPS and GIS to apply removal efforts at the landscape level. Residual animals were removed using dogs and targeted trapping. New techniques for detecting nutria at low densities are currently being evaluated including dogs, lures and attractants, call surveys, judas nutria, and decoy cages. Recovery of nutria-damaged marsh has been significant and has halted further conversion of marsh to open water. The programme now aims to create a nutria-free coastal marsh ecosystem across the Delmarva Peninsula by 2014.

Keywords: Coypu, impacts, eradication, Chesapeake Bay, Delmarva Peninsula, trapping

INTRODUCTION

The eradication of invasive pests is increasingly being attempted by conservation managers while the size and complexity of successful eradications has surpassed what was previously considered feasible (Donlan *et al*. 2003). Feral pigs and goats have been eradicated from several large islands in the Galapagos (Cruz *et al*. 2005; Campbell and Donlan 2005) and the size of New Zealand Islands from which Norway rats have been successfully eradicated has increased logarithmically (Clout and Veitch 2002).

The Delmarva Peninsula, which is bordered by the Chesapeake and Delaware Bays and the Atlantic Ocean, comprises the state of Delaware and parts of Maryland and Virginia (Fig. 1). The peninsula supports tidal wetland habitats recognised as among the most important in the United States and as "Wetlands of International Importance" under the Ramsar Convention Treaty (Tiner and Burke 1995). The wetlands are home to numerous fish and wildlife species, and support commercial and recreational fishing, hunting, trapping, bird watching, wildlife viewing, and photography.

Nutria (*Myocastor coypus*), a tropical, aquatic South American rodent, was introduced to the United States in California in 1899 and to southern states in the early 20th Century for fur farming and weed control (Evans 1970; Willner *et al* 1979; LeBlanc 1994; Hess *et al*. 1997). After their introduction to Delmarva Peninsula in 1943, numbers of nutria increased to at least 50,000 in the early 1990s (Carowan pers. comm.). In the Delmarva marshes, nutria mostly feed on the roots of Olney three-square bulrush (*Scirpus olneyi*), a native emergent grass that grows 1-1.5 meters above water and supports a submersed root mat in highly erodible sediment. When nutria excavate roots, they expose the sediment to tidal erosion and brackish wetlands to salt water intrusion (Haramis and Colona, unpublished). Wetlands are converted to open water, removing all habitat benefits of the marsh for native species. On the Blackwater National Wildlife Refuge (CMNWRC), for example, nutria destroyed more than half of its original marsh (2833 ha).

Efforts to control nutria on Delmarva through commercial and recreational trapping did not prevent damage to three-square bulrush marsh. Maryland officials then consulted Dr. L.M. Gosling who, after several decades of research, failed attempts and effective trials, led a team of 24 trappers to successfully eradicate nutria from Britain over six years in the 1980s (Gosling 1989). Based on Gosling's recommendations, the task force focused on eradication as the primary strategy for restoring and protecting nutria-damaged marshlands in the Chesapeake Bay. Systematic trapping was identified as the primary method for reducing nutria populations.

In 1997, a partnership of federal and state agencies and private interests was formed to develop and implement a pilot project with the ultimate goal of eradicating nutria on Maryland's Eastern Shore. The Nutria Control/Marsh Restoration Pilot Project aimed to gather data on the population of nutria in CMNWRC, Fishing Bay Wildlife Management Area (FBWMA), and Tudor Farms and adjacent properties in Dorchester County. Information on nutria population size, physiology, reproduction, behaviour, and movement were used to develop and test trapping strategies to maximise removal. Two years were dedicated to the collection of baseline data (Phase I) and four years (2002-2006) to test and implement eradication strategies on the 24,300 ha encompassed by these areas (Phase II). In 2007, trapping of nutria began in neighbouring counties and the eradication zone was redefined to include all of Delmarva Peninsula. Although not an island per se, the peninsula is sufficiently isolated from mainland nutria populations that the risk of recolonisation through immigration is thought to be near zero.

This paper describes the methods used to reduce nutria populations to near zero densities from 2002- 2009 as part of a campaign to eradicate the species from the Delmarva Peninsula.

MATERIALS AND METHODS

Project management and staffing

An eight member management team of senior-level representatives from U.S. Fish and Wildlife Service, U.S. Department of Agriculture (USDA), Maryland Department of Natural Resources (MDNR), U.S. Geological Survey (USGS), and Tudor Farms oversaw the project and was primarily responsible for securing funding, obtaining political support, and providing technical support to field operations. A full-time wildlife biologist managed operations and supervised staff members, which included 17 full-time wildlife trapping specialists, one full-time maintenance worker who maintained vehicles, boats and trapping equipment, and a part-time administrative assistant.

Pages 313-319 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

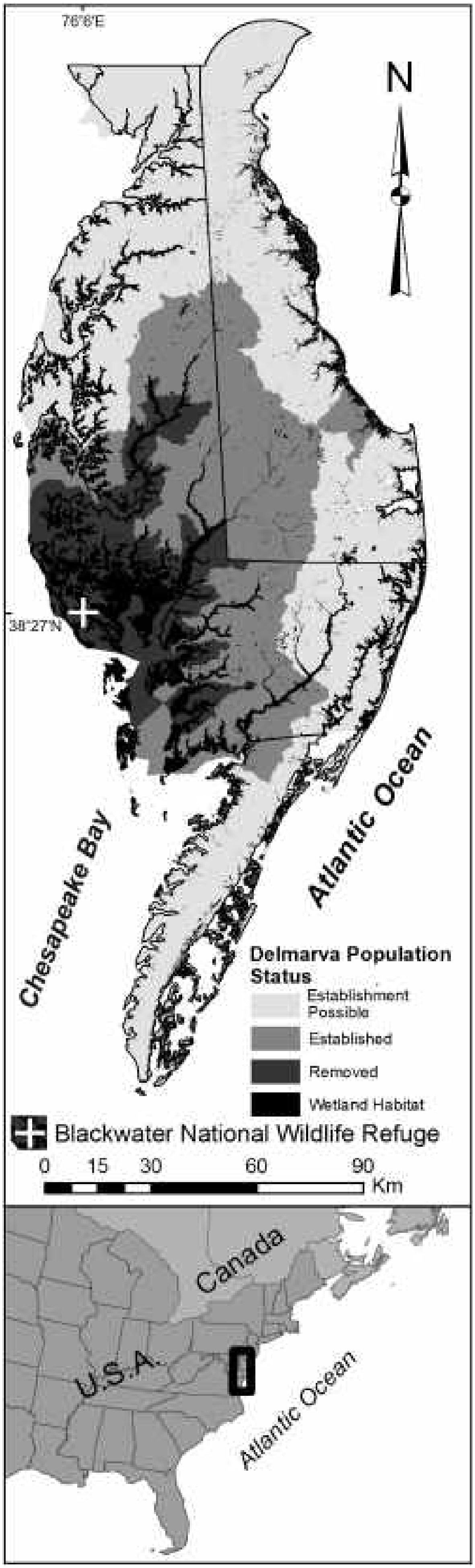

Fig. 1 Distribution of wetland habitats on Delmarva Peninsula and population status by subwatershed in May, 2011.

Phases of Eradication

Our nutria eradication campaign can be broken into six phases:

1) Survey: Define the distribution of nutria on the Delmarva Peninsula.
2) Knock-down: Rapid depopulation of metapopulations identified in the survey phase.
3) Mop-up: Targeted removal of residual nutria remaining after the knock-down phase is completed.
4) Verification: Population monitoring to confirm that eradication at the management unit level was successful.
5) Surveillance: Continued monitoring at the landscape level.
6) Biosecurity: Implementation of strategies to prevent the reinvasion of nutria.

While the process outlined above was generally followed sequentially, we were frequently engaged in multiple phases simultaneously in different management units. In addition, the progression between phases was not always linear and the transition between phases was not always discrete.

Removal methods

Nutria were primarily removed through trapping, hunting and shooting. Trap devices used included rotating-jawed body-gripping traps (Conibear type) (Fig. 2), foothold traps (Fig. 3), cage/box traps, and cable restraining devices (snares). Traps were set on nutria trails, in ditches, along waterways and at approaches to natural and artificial (false) nutria beds and haul-outs, on floating support frames, and floating platforms. Methods used included: 1) "blind" sets in natural travel ways; and 2) lured sets using urine collected from captive animals, scats, anal gland lure, disturbed earth, and cut vegetation. Traps were typically set on sign of nutria presence. In low density areas, where nutria are more difficult to detect, trapping specialists used their understanding of nutria behaviour and movement to place sets where they were most likely to capture nutria

Fig. 2 A 17.8 cm body-gripping (Conibear) trap set on a floating platform. The trap triggers are spread to allow smaller non-target species to pass through the trap.

moving through the area. Kill traps were checked within 96 hrs and live traps within 24 hrs. Non-target captures of native mammals, birds and reptiles were minimised by manipulating trap trigger and pan configurations, placing jump sticks or obstructions to block non-target access to traps, and selectively avoiding areas used by non-target species.

Hunting and shooting using small calibre rifles, shotguns, and handguns, was conducted year round, but was most effective in winter when marshes and waterways froze and reduced escape routes for nutria and snow cover provided a tracking substrate. Trained dogs were used throughout the year to detect and remove nutria, particularly in previously trapped areas.

Use of toxicants (e.g., zinc phosphide) was considered during the planning phase of the programme, but rejected because of concern over potential non-target impacts. The high success of nutria removal through trapping and hunting, followed by spot removal using detection dogs, has so far precluded any need to use toxicants.

Initial Population Reduction Strategies

There are almost 200,000 ha of wetland habitats on the Delmarva Peninsula, which required a systematic trapping programme in manageable trapping units. A Geographic Information System (GIS) was used to overlay a 402 m x 402 m rectangular grid of trapping units on a wetland map of the Delmarva Peninsula. Two removal strategies were implemented based on the spatial distribution of marsh habitat.

Fig. 3 A foothold trap set on an imitation nutria bed. The trap is wired to a one-way slide lock attached to a cable anchored in deep water. This submersion set is designed to quickly drown captured nutria. Bamboo poles are placed to reduce non target bird captures.

First, progressive sweeps were used in large contiguous blocks of marsh habitat. A continuous band of trapping units was established across the marsh, bridging non-nutria habitat (uplands or open water) on either side. Trapping specialists used handheld GPS receivers to ensure that they were trapping assigned units. As nutria in each band of trapping units were reduced to very low density, trappers moved forward to the next un-trapped unit. When capture rates in a trapping unit slowed, traps were established in the next adjacent trapping unit, leaving some traps behind to capture animals attempting to penetrate the trapping front. A swath of continuous trapping activity was thus spread across the marsh, three to four trapping units wide, with trapping intensity highest at the leading edge.

Second, a simultaneous blitz removal strategy was used in smaller, isolated marshes that could be trapped as a single unit. Such marshes typically bordered rivers throughout their tidal reach. Trappers were assigned to each section of river frontage and all marsh units were trapped simultaneously.

Trapping units were considered as depopulated after two weeks without a nutria capture. Data collected included the number of trap nights, the location, age, and sex of each nutria removed, and the identity and location of all non-target captures.

Hunting and shooting were used extensively during winter, when freezing conditions impeded trapping efforts and often caused nutria to aggregate. Areas that were heavily hunted were subsequently trapped once weather conditions permitted.

Monitoring

Following initial knock-down, trapping units were monitored every 3-12 months, depending on access and risk of reinvasion, for signs of nutria activity using: 1) intensive ground or shoreline searches documented with GPS tracks; 2) searches with dogs trained to find nutria; and 3) surveys of nutria sign at false beds. In order to reduce non-target impacts, traps were not used as monitoring devices unless sign was detected. Nutria population status was assigned to one of three categories for each trapping unit surveyed:

Resident: Evidence of occupancy including well-used nutria trails, bedding and feeding activity and/or the presence of multiple sizes of fresh scats indicating the presence of different age groups of nutria. Set traps would have a high probability of capture.

Transient: Evidence that a nutria passed through, but was not inhabiting the area. Usually a lone set of tracks or small amounts of scat of indeterminate age would be classified as transient. Set traps would have a low probability of capture.

Absent: No evidence of nutria detected.

With increasing size of the eradication zone, monitoring effort in previously trapped areas increased proportionately and competed directly with efforts to expand knock down efforts into new areas. In order to manage these competing needs, monitoring areas were prioritised for survey based on their risk of re-infestation as determined by prior occupancy, proximity to un-trapped areas, or presence of preferred habitat. High priority trapping units were monitored with increased frequency until failure to detect nutria after repeated visits warranted a reduction in priority. Mop-up trapping efforts were initiated upon the discovery of resident sign and discontinued after two weeks without a capture and failure to detect fresh sign.

Table 1 Total wetland area (ha) in Maryland counties and areas subject to nutria control in 2003-2008. No nutria control was conducted in Queen Anne, Kent, Cecil, and Worchester Counties (29,520 ha of marsh) and no new area received treatment in 2009.

County	Avail. wetland	2003	2004	2005	2006	2007	2008	Area trapped	Percent available
Dorchester	54,628	11,738	11,798	6607	10,254	2248	253	42,897	79%
Somerset	42,715	0	0	0	0	6833	2901	9734	23%
Wicomico	13,272	0	0	0	0	5473	0	5473	41%
Talbot	5122	0	0	0	0	0	1482	1482	29%
Total	118,448	11,738	11,798	6607	10,254	14,554	5407	60,358	41%

Table 2 Number of nutria removed and percent of first year removal from Initial Knock-down Areas (IKDAs) during eradication efforts on Delmarva Peninsula.

IKDA	2003	2004	2005	2006	2007	2008	2009	Total
2003	4795	370	127	70	16	19	5	5402
%	100%	7.7%	2.6%	1.4%	0.3%	0.4%	0.1%	
2004		3071	290	63	20	41	4	3489
%		100%	9.4%	2.1%	0.7%	1.3%	0.1%	
2005			677	108	17	127	69	998
%			100%	15.9%	2.5%	18.8%	10.2%	
2006				318	32	22	9	381
%				100%	10.1%	6.9%	2.8%	
2007					812	79	88	979
%					100%	9.7%	10.8%	
2008						1183	387	1570
%						100%	32.7%	
Total	4795	3441	1094	559	897	1471	562	12819

Table 3 Time required to achieve an approximate 100% reduction in nutria numbers in trapping units during initial trap out, and number of nutria removed. Data based on IKDAs trapped in 2003-2008.

	Trapping units reduced to near-zero density			Nutria Removed		
Week	Number	%	Cumulative %	Number	%	Cumulative %
1	0	0.0	0.0	4584	51.1	51.1
2	145	11.4	11.4	1779	19.9	71.0
3	208	16.3	27.7	837	9.3	80.3
4	176	13.8	41.5	447	5.0	85.3
5	177	13.9	55.4	303	3.4	88.7
6	153	12.0	67.4	247	2.8	91.5
7	99	7.8	75.1	148	1.7	93.1
8	63	4.9	80.1	148	1.7	94.8
9	44	3.5	83.5	70	0.8	95.5
10	38	3.0	86.5	64	0.7	96.3
11	23	1.8	88.3	45	0.5	96.8
12	18	1.4	89.7	28	0.3	97.1
13-30	131	10.3	100.0	262	2.9	100.0
Total	1275			8962		

Analysis

We tallied the amount of effort required to reduce the nutria population to near-zero by counting the number of weeks of trapping required and back-calculating the percentage of the pre-existing population captured during each week of trapping, accepting that this slightly overestimates percentage removed as an unknown number of nutria remained un-trapped. By determining the total number of nutria removed from a trapping unit during initial removal and dividing that number into the weekly capture total, we were able to determine the percentage of the presumed population that was taken during each successive week of trapping.

Initial knock-down areas (IKDAs) were defined by the year in which knock down activities were initiated and the area covered in that year. We determined the number of nutria removed from each IKDA during the year of initiation and compared the number of nutria removed during mop-up efforts in the same areas in subsequent years. Traps were only set when sign was detected during monitoring, thus trapping effort was not applied equally across years and catch per unit effort data was not compared. However, the reduction in number of nutria removed was evaluated to gauge the magnitude of the population reduction.

Table 4 Number and percent of nutria removed by method during initial population reduction and clean-up phases of eradication.

	Knock-down		Mop-up		Total	
Method	Number	%	Number	%	Number	%
Conibear	7457	67.9%	762	42.4%	8219	64.3%
Shooting	1316	12.0%	101	5.6%	1417	11.1%
Submersion foothold	927	8.4%	449	25.0%	1376	10.8%
Dog	470	4.3%	344	19.2%	814	6.4%
Foothold	460	4.2%	78	4.3%	538	4.2%
Snare	105	1.0%	13	0.7%	118	0.9%
Floating Conibear	97	0.9%	10	0.6%	107	0.8%
Hand caught	66	0.6%	15	0.8%	81	0.6%
Platform Trap (foothold)	62	0.6%	18	1.0%	80	0.6%
Platform (conibear)	15	0.1%	2	0.1%	17	0.1%
Cage	8	0.1%	4	0.2%	12	0.1%
Spotlight/shoot	6	0.1%		0.0%	6	0.1%
Grand Total	10,989	100 %	1796	100 %	12,785	100 %

Table 5 Trap nights and catch per unit effort (nutria/1000 trap nights) for top three trapping methods and total captures using non-trapping methods during initial knock-down and mop-up during eradication efforts on Delmarva, 2002-2008.

	Initial Knock-down			Mop-up		
Method	Trap nights	Captures	CUE	Trap nights	Captures	CUE
Body-grip	602,636	7462	12.38	56,917	746	13.11
Submersion	36,538	928	25.39	17,356	434	25.01
Foothold	13,160	460	34.9	1960	78	39.79
Shooting	n/a	1316	n/a	n/a	1417	n/a
Dog	n/a	470	n/a	n/a	814	n/a

RESULTS

Between 2003 and 2008, the campaign against nutria was conducted over nearly 61,000 ha of the 148,000 ha wetland habitat on Maryland's eastern shore, as determined from National Wetland Inventory maps (Table 1). Knock-down activities were initiated on new areas each year until 2009, when verification and mop-up activities left little time for expansion into new areas. Nutria catches on IKDAs were used to track progress in population reduction (Table 2). In the third year following initial knock-down, mop-up efforts yielded <3% of the population removed in the initial year of treatment for IDKAs 2003-2006. The exception was IKDA 2005, where an area was not trapped until 2008 due to access restrictions imposed by a private landowner (Table 2). More than 100 nutria were removed from this property. In fact, many of the nutria captured in 2003 and 2004 IKDAs during monitoring were trapped within 13 km of this property, well within dispersal distances observed by GPS/radio-tagged nutria released as part of an ongoing Judas experiment (not reported here).

Nutria were encountered in approximately one third of the trapping units inspected and were reduced to very low numbers in 75 % of those within seven weeks of trapping (Table 3). A few units required up to 30 weeks to capture the last one or two nutria. Typically, more than half of the original population was captured in the first week of trapping, 80% by the end of the third week, and more than 90 % by the end of the sixth week of trapping. In many trapping units, catching the last 5-10% of the population took as long as or longer than capturing the first 90-95 %.

The most productive methods of nutria removal during the initial depopulation phase were body-gripping traps, shooting, footholds set on submersion cables, dogs, and staked foothold traps (Table 4). Staff accumulated 652,334 and 76,233 trap nights during knockdown and mop-up trapping efforts, respectively. Body gripping traps accounted for 92 % of trap nights and 84% of captures during knock-down trapping and 59% of trap nights and 75% of captures during mop-up trapping. Submersion footholds accounted for 6% of trap nights and 10% of captures during knock-down, but 23% of trap nights and 35% percent of captures during mop-up trapping. Staked footholds accounted for 2% of trap nights and 6% of captures during both knock-down and mop-up trapping phases. During initial knockdown, catch rates were lowest for body-gripping traps and highest for staked foothold traps. These latter were marginally more effective during mop-up trapping (Table 5).

Populations that remained or developed after initial population reduction typically comprised small groups ranging in size from two to six animals, although one group of 41 animals eluded detection for three years. Analysis of the sex and age distribution of the captured nutria led us to conclude that this abnormal population arose from a small group of three to six females that immigrated sometime during the third year following initial knock-down.

DISCUSSION

We implemented a systematic hunting and trapping programme that effectively reduced feral nutria populations within 16 ha trapping units to near zero within four to eight weeks per unit. Progressive and sequential treatment of trapping units across larger management units (watersheds) enabled us to effectively eliminate nutria over >60,000 ha of sensitive coastal wetlands in the Chesapeake Bay Watershed. Several mop-up sessions have been applied throughout this area, much of which is now in the verification phase. Nutria have not been detected in some watersheds for several years and these sites are now in the surveillance phase.

Although the same removal methods were used during knock-down and mop-up trapping phases, the relative importance of different trapping techniques was influenced by the needs of knock-down versus mop-up trapping strategies. For example, body-gripping traps accounted for the largest number of animals in both phases, but submersion footholds and detector dogs played a greater role in removal during mop-up efforts. One possible explanation for the increased importance of submersion footholds is that nutria at low densities move greater distances along waterways in search of other nutria and are therefore more vulnerable to footholds set at false beds along waterways. In addition, specialists aided by dogs are more efficient at finding nutria in areas of low density than specialists without dogs. We thus relied heavily on detection dogs during mop-up phases.

In England, catch per unit effort was used to indicate declines in population (Gosling and Baker 1987), but we did not detect significant changes in catch per unit effort between knock-down and mop-up trapping phases. Furthermore, box traps were used in England to allow the release of non-target species and a consistent trapping effort during consecutive trapping sessions. However, we set kill traps only where evidence of nutria was documented during intensive sign searches. This targeted approach to removing residual populations enabled us to reduce impacts to non-target species by restricting trapping to areas occupied by nutria. Compared with experiences in England, our approach required a greater investment in alternative detection methods.

Differences were recorded in the catch per unit effort of body-gripping versus foothold traps is likely due to the way in which traps are set. Body-gripping traps are often set as blind trail sets in higher trap densities to cover the myriad of trails available. Footholds, in contrast, are most often set selectively along waterways in conjunction with a false bed and/or urine or other visual or olfactory attractant. The difference between submersion and staked foothold efficiency is probably due to small sample sizes and the fact that staked footholds were only used during the first few months of knock-down trapping. The use of staked footholds was largely discontinued after submersion sets were approved as a lethal trapping technique, allowing us to increase trap check intervals from 24 to 96 hours.

Monitoring the previously trapped populations remained one of the programmes biggest challenges. With 61,000 ha of depopulated habitat spread across five counties, returning to these areas on a regular basis required an exhaustive effort that precluded expansion into new areas. Yet, expansion into new areas was necessary to reduce the risk of reinvasion of the nutria-free zone. Thus, these priorities competed for limited staff resources and time. Additionally, many private landowners continued to restrict our access during the non-growing season, from September to the end of January, primarily because of recreational hunting.

Damaged marshes often recovered rapidly after nutria were removed. As nutria populations approached zero, staff reported that nutria swim channels were reclaimed by rhizome growth from three square bulrush. The resulting network of new roots trapped sediments that filled in swim channels, thereby eliminating the primary route of erosion for organic soils dislodged by nutria foraging habits. These anecdotal observations were corroborated by quantitative vegetation studies conducted at Patuxent Wildlife Research Center, which showed a dramatic recovery in areas extensively damaged by nutria (e.g., Figs 4a, b; Haramis *et al*. 2006).

This project was the first large scale attempt to eradicate nutria in North America. The type and distribution of habitat on Delmarva differs significantly from nutria habitat in England. While the UK example provided valuable insights, the political, social, and ecological conditions dictated a different approach in Delmarva and yielded new lessons including:

1) Eradication is achievable at the trapping unit level when integrated methods are applied systematically by skilled technicians. By replicating the process progressively across management units nutria densities were reduced to near zero at the landscape level.

Fig. 4 (A) A wildlife specialist examines a nutria eat out in Monie Bay watershed, Somerset County, Maryland in May 2007. (B) the same marsh in May 2009, during the second growing season following eradication of nutria.

2) Cooperation of private landowners is important to putting every nutria at risk, although it is likely that nutria residing in relatively small private in holdings can be trapped from the periphery.
3) Techniques used effectively during the knock-down phase may not be sufficient to achieve final eradication once the population has been reduced to extremely low densities.
4) Staff must be prepared to develop and adapt tactics and strategies when new challenges reveal themselves.
5) Efficiency varies seasonally. Nutria are more difficult to detect during the summer months when lush vegetation conceals evidence of occupancy and nutria movements appear to be minimal. Conversely, late fall through early spring is an optimal period for detecting nutria as vegetation dies back and nutria are more active.
6) Nutria may restrict activity or abandon sites subjected to intense daily human activity. Reducing the frequency of trap checks to 96 hours appeared to reduce incidence of site abandonment.

CONCLUSIONS

The Chesapeake Bay Nutria Eradication Program now aims to create a nutria-free coastal marsh ecosystem across Delmarva Peninsula by 2014. Given the worldwide distribution of nutria and its status as an invasive pest (Carter and Leonard 2002), the lessons learned from our programme will help instruct those interested in controlling or eradicating nutria elsewhere. Ongoing control programmes in Italy and Louisiana, USA, show promise for reducing damage to acceptable levels if eradication is deemed impossible (Bertolino and Viterbi 2009, Wiebe and Mouton 2009). The Delmarva programme has important implications for enhancing the effectiveness of control efforts, identifying additional eradication opportunities, and preventing invasion through the early detection and removal of invaders.

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Invasive species removal and ecosystem recovery in the Mariana Islands; challenges and outcomes on Sarigan and Anatahan

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Abstract Sarigan Island had a successful eradication of pigs (*Sus scrofa*) and goats (*Capra hircus*) in 1998. Following the removal of these species, native forest became blanketed by the invasive vine *Operculina ventricosa*. Subsequently, the cover from *O. ventricosa* has stabilized possibly due to competition with two other exotic vine species, drought, and the effects of storms. Native forest cover has increased greatly. Most species of flora and fauna have increased in abundance with native snails and skinks showing some of the greatest gains. Success of the Sarigan ungulate eradication and subsequent responses by native species prompted an attempt to eradicate pigs and goats on nearby Anatahan Island. Support of the project by local inhabitants was gained through education and incentives. Soon after the eradication began, a catastrophic volcanic eruption destroyed the local village and most populations of native species on the island as well as the remaining goats. However, small numbers of other introduced vertebrate species survived. Anatahan Island is still blanketed by ash, is in very early stages of re-vegetation and may one day join Sarigan as a site for bird introductions.

Keywords: *Capra hircus*, eradication, megapode, *Operculina ventricosa*, *Partula gibba*, recovery, *Sus scrofa*, volcano

INTRODUCTION

Sarigan and Anatahan Islands are two of the fourteen islands that make up the United States possession of the Commonwealth of the Northern Mariana Islands (CNMI) (Fig. 1). The nine northern-most islands of the CNMI are mostly active volcanoes. The islands are typically steep-sided cones rising abruptly out of the ocean and inhabited by fewer species of flora and fauna than the six limestone islands to their south, the sixth island being the US Territory of Guam. Past attempts to populate or otherwise economically utilise the northern islands have met with failure due mainly to volcanic activity, severe typhoons, and difficult logistics. Unfortunately, a remnant of these attempts has been an abundance of feral goats (*Capra hircus*) and pigs (*Sus scrofa*) on Sarigan, Anatahan, Alamagan, Pagan, and Agrihan, which compose 87% of the landmass of the nine islands. In addition, Pagan and Alamagan have feral cattle (*Bos taurus*). Other pest species of concern include: cats (*Felis catus*) on Anatahan, Sarigan, Alamagan, Pagan, and Agrihan; dogs (*Canis familiaris*) on Agrihan; and rats on all islands (*pers. obs*.).

Eradication of pigs and goats have been completed or attempted on Sarigan and Anatahan Islands. In this paper, I describe the methods used on Anatahan and the outcomes recorded after the campaigns on both islands.

The U.S. Fish and Wildlife Service (FWS) Biological Opinion (6 April 1998) recommended that the Navy fund conservation and recovery projects in the Marianas to improve the habitat and population size of the federally listed Micronesian megapode (*Megapodius laperouse laperouse*) as mitigation for bombing activities on Farallon de Medinilla. To date, the Navy has provided approximately $750,000 in funding for baseline studies and the removal of feral ungulates on Anatahan for habitat restoration. However, no funds were allocated to the removal of other invasive mammals such as cats and rats. The ungulate project is a cooperative effort by FWS, Navy, CNMI-Division of Fish & Wildlife (DFW), and the Northern Islands Mayor's Office (NIMO).

STUDY SITES

Sarigan (16º 42'N 145º 46'E) is a 500 ha island about 195 km north of Saipan. Over 100 years of grazing by feral ungulates had left patches of bare ground, practically no forest understory, and dry remnant native forest progressively being replaced by introduced grasslands dominated by golden beargrass (*Chrysopogon aciculatus*). In 1998, feral goats and pigs were eradicated and within six months there was extensive colonisation by the invasive vine paper rose (*Operculina ventricosa)* (Kessler 2002). Other species with increased population sizes (detailed elsewhere in the paper) included native skinks, birds, and native tree snails. An additional result of the eradication is that Sarigan has been chosen as the first island for the translocation of bird species from the southern islands as a precaution against future establishment of the brown tree snake (*Boiga irregularis*). This snake is infamous for its role in the extinction of Guam's avifauna (Savidge 1987; Fritts and Rodda 1998).

The success of Sarigan's ungulate eradication prompted a similar attempt on Anatahan Island (16º 21'N 145º 41'E), 40 km further south. Anatahan is 3200 ha (9 km by 4 km) and rises to 788 m. It has two volcanic craters; the older centre crater forms a vegetated central basin. The smaller eastern crater was characterised by steep vertical slopes with some vegetation and bubbling mud pits at the base. On the lower coastal slopes, *Cocos nucifera* was managed as a copra plantation from 1900 – 1940 (Fritz 1902; Ohba 1994). Native forest on the steep side slopes is characterised by tropical almond (*Terminalia catappa*) (Ohba 1994). Toward the tops of the slopes is swordgrass (*Miscanthus floridulus*) or *Chrysopogon aciculatus* grasslands, with the endemic giant tree fern *Cyathea aramaganensis* where fog conditions exist (Ohba 1994). Much of the native forest had been severely degraded by feral goats and pigs (Pratt and Lemke 1984; Reichel 1988; Rice 1992; Ohba 1994; Kessler 1996), leaving many areas of patchy forest with little to no ground cover and large areas of easily erodible loose soil.

Pigs were already established on Anatahan during the late 1890s (Fritz 1902) and goats are thought to have been introduced in about 1960 (Reichel *et al*. 1988). The pigs mainly preferred the coconut forests, level areas, and those areas that had some standing fresh water. Goats were found throughout and had severe effects on all vegetation types. The extent of forest reduction (60% on the south side) can be observed by comparing aerial photographs taken in 1944 and with those repeated in 1999 (Kessler 2000). In 1995 the goat population was estimated at 5000 to 6000 animals (Worthington *et al*. 2001).

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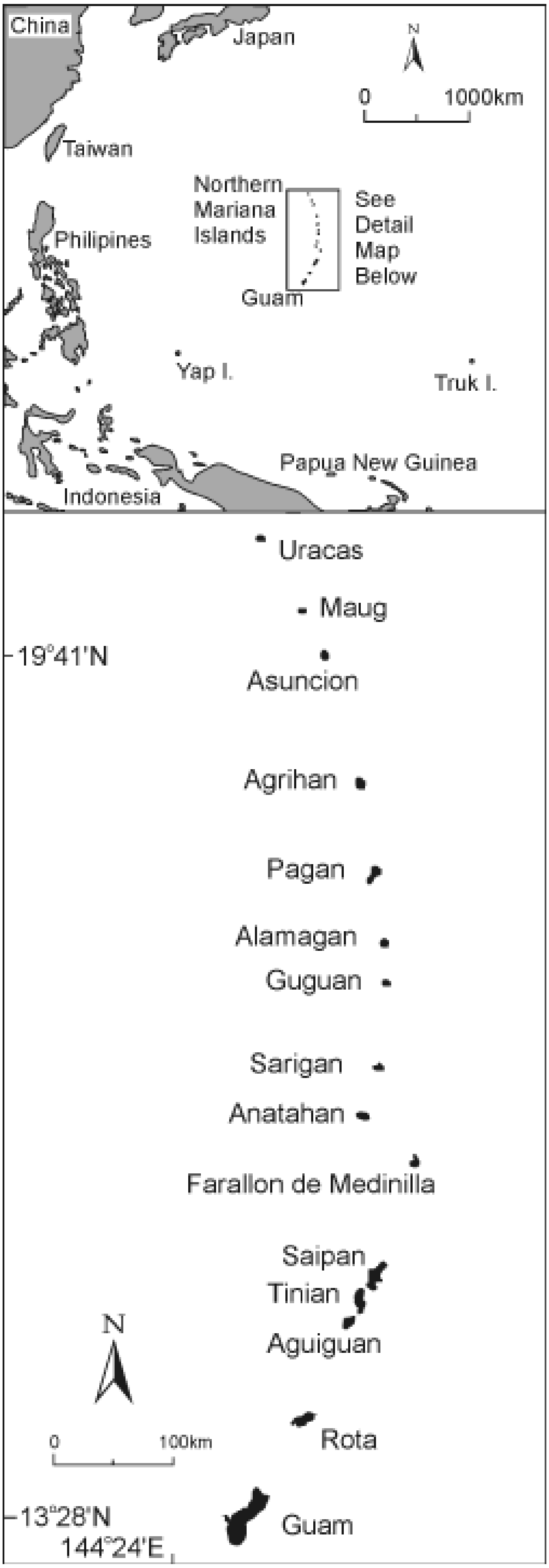

Fig. 1 The US territory of Guam and the 14 islands that make up the US Commonwealth of the Northern Mariana Islands.

METHODS

Methods and results for the eradication campaign on Sarigan were described in Kessler (2002).

Wildlife and vegetation surveys on the island have been undertaken since 1997 when baseline data were collected before the 1998 ungulate eradication. These expeditions were conducted by the CNMI-DFW in 1997, 1999, 2000, and 2006 and include data on changes to vegetation, reptiles, birds, and snails (Fancy *et al*. 1999; Morton 2000; CNMI-DFW 2000, 2008).

The Anatahan ungulate removal project was modelled after the Sarigan project and was divided into five phases: 1- Reconnaissance and survey; 2 - Base camp establishment; 3 - Shooting programme; 4 - Removal of remnant population/individuals; 5 - Follow-up monitoring and re-surveying.

Phase 1 started in 1997, with the assessment of the ungulate populations and the establishment of vegetation photo plots, and continued in May 2002 with baseline surveys. Also in May 2002, Phases 2 and 3 began with the transportation of supplies and two weeks of shooting. In January 2003, aerial hunting began on 75% of the island and was to have continued on a monthly schedule. As part of the hunting programme, eight female goats were captured, fitted with radio telemetry collars as "Judas" goats (Taylor and Katahira 1988), and released into their original home ranges.

Initial flora and fauna baseline surveys were completed in May 2002 by CNMI-DFW. The two-week initial ungulate shoot was designed to slow forest collapse and was estimated to have removed half of the goat population. Because feral cats were also present, attempts were made to remove as many as possible during ungulate shooting operations. An agreement made with NIMO required a fence to be built that would constrain ungulates to 25% of the island for use by villagers. Phase 4, which involved ground hunting using eight to ten hunters with dogs, along with a separate crew of fence builders, was organised to begin in June 2003.

In May 2003, without prior warning, the island's volcano erupted for the first time in recorded history (Truesdall *et al*. 2005). Eruptions continued sporadically for over two years. The final eruptive episode covered the island in ash, which removed 98% of the vegetation and extirpated all land birds.

After the initial eruption in May 2003, the Governor of the CNMI through the Emergency Management Office (EMO), restricted travel to scientific expeditions, prohibited entry to the village and cancelled construction of the fence. Due to the volcanic hazards, EMO has restricted time spent on the ground and limited eradication operations to aerial hunting. Aerial operations consisted of one trip per month (volcanic conditions permitting) lasting two days. On average, this enabled 12 hours of flight time per month with about four hours of actual aerial searching per day (the additional hours being used for transport). However, a continuous monthly schedule has never been achieved due to environmental conditions, logistics (all fuel must be prepositioned), and funding delays.

Aerial shooting involved two hunters and the pilot. One shooter was assigned responsibility to tally kills and record areas searched. Helicopter shooting usually took place in the last hours of the day, but was initially varied to determine the most productive times. One hunter used a 12 gauge shotgun, shooting three inch shells with double-O buckshot and sat behind the pilot. The other hunter was opposite the first and used a semi-automatic .223 calibre rifle with telescope sight and a bullet catcher. This arrangement allowed the pilot to use either side depending on the winds and terrain. The shotgun was used for calm conditions and getting close to targets and the rifle was used in windy conditions that required shooting from a distance.

Before the eruption, a helicopter was used to transport hunters to the ridge tops. Personnel then moved in a line down slope keeping in close contact by radio and/or sight to ensure total coverage of each section. Ground parties then assembled at a collection point on the coast and were picked-up and transported by small boat to a location that could be accessed by the helicopter. This routine could be done twice daily. During the May 2002 operation, an effort was made to salvage goat and pig carcasses and transport them to the village. Carcasses were dragged to a central location by the hunters, roped together in groups not weighing more than the helicopter's capacity (~ 300 kg), and slung to the village to be processed by local inhabitants. Freezers and generators were supplied for storage of the meat.

A final hunt was scheduled to include ground hunting with dogs.

RESULTS

Sarigan

Vegetation

Tree and herbaceous species have quickly filled in open areas and the island is no longer an open forest without understory and areas of grassy fields. Now there is a tangled jungle, closed canopy, and 100% ground cover in areas without trees. Areas once covered with grass are now studded with saplings and covered with vines. The total number of tree species identified in the forested areas has increased between surveys and the overall density of tree species has increased more than tenfold from 1.48 trees/100m^2 in 1999 to 13.70 trees/100m^2 in 2006 (CNMI-DFW 2008). The average canopy cover for all forest transects in 2006 was 77%, an approximate 20% increase from 2000 when overall forest canopy cover was 52% for all transects. The range of canopy cover for forest transects in 2000 was 49% to 76% and in 2006 it was 72% to 92%. Canopy cover on a grassy field transect went from 0.4% to 15% (CNMI-DFW 2008).

Wildlife

Native arboreal snail populations on Sarigan were most dense in forested areas dominated by broadleafed native trees. At one station in the native forest, 448 specimens of *Partula gibba* and 204 of *Succinea* sp. were encountered within a 25 m^2 quadrant. (CNMI-DFW 2008).

Three species of skinks were recorded on Sarigan in 1997: snake-eyed skink (*Cryptoblepharus poecilopleurus*), blue tailed skink (*Emoia caeruleocauda*), and Slevin's skink (*Emoia slevini*). Subsequently, catch rates for *E. caeruleocauda* increased dramatically and peaked in 2000, then fell slightly in 2006, but were still above catch rates for 1997. Catch rates of the endemic *E. slevini* also increased, particularly in native forest. In contrast, catch rates of *C. poecilopleurus* have rapidly declined. This species was found only in the native forest in 2000, but was not captured at all in 2006 (CNMI-DFW 2008).

In 1997, five species of land birds inhabited Sarigan: the endangered Micronesian starling (*Aplonis opaca*), white-throated ground dove (*Gallicolumba xanthonura*), Micronesian megapode (*Megapodus laperouse*), Micronesian honeyeater (*Myzomela rubratra*), and collared kingfisher (*Todirhamphus chloris*). Megapodes and honeyeaters are the two most frequently detected species on Sarigan. Detection rates have increased for both species, but the greatest increase has been for honeyeaters. The starlings and kingfishers have declined slightly in detection frequency, whereas the white-throated ground dove has shown no trend and remains elusive (CNMI-DFW 2008).

Two species have been added to the avifauna of Sarigan. The Marianas fruit dove (*Ptilinopus roseicapilla*), is apparently a natural colonisation. The bridled white-eye (*Zosterops conspicillatus saypani*) was purposefully introduced in 2009 to expand its range and reduce the risk of extinction if brown tree snakes become established in the southern islands of the CNMI.

Anatahan

Eradication

In May 2002, the initial shoot from helicopters removed 1740 goats, 32 pigs, and five cats over 31.5 hunting hours. The highest one-day kill rate for was 106 goats/hour, while the overall average was 55 goats/hour. Concurrently, the ground crew removed 681 goats, 30 pigs and one cat in approximately 344 man-hours and with two hours of helicopter transport time. The combined aerial and ground shooting total over 14 days was 2421 goats, 62 pigs, and six cats.

In January 2003, the eight "Judas" goats with radio collars were deployed around the island, following which an additional 144 goats and one pig were removed in six hours of aerial hunting. Ground crews removed an additional 40 goats and five pigs over approximately 75 man-hours.

During the pre-eruption period of January through April 2003, while using "Judas" goats, 784 goats, 47 pigs, and one cat were removed during 30 aerial hunting hours. The highest one-day kill rate from the helicopter dropped to about 40 goats/hour and the average was about 25 goats/hour.

During the two years of active eruptions, there was some limited aerial shooting as conditions permitted. However, once activity ceased (December 2005), no goats were found and they are now considered eradicated. Some pigs had persisted with another 18 removed through aerial hunting. By January 2010, only three pigs were estimated to have survived. Only one cat was removed during this period.

Meat Salvage

About 50 goats and five pigs were moved to the village after helicopter recovery before villagers become overwhelmed by the processing effort and the transfers were stopped. Approximately two hours of flight time was wasted at US$1200/hr in addition to the cost of two generators and freezers and the field time of six staff. Although the task of saving meat was overwhelmed by the physical effort required, it did stop the complaints about "wasted meat." A greater number of skinners with better skills and determination might have yielded different results. However, the fact that most goats were shot in extreme terrain would have limited the salvage to < 300 animals.

DISCUSSION

Sarigan

Based on survey results, the trend of increasing tree species richness and density should continue as species sighted, but not yet detected on transects, become more established. Forest composition is changing toward a more native and diverse ecosystem and areas of bare soil now have ground cover. Introduced short grasses are declining in extent as they are replaced by forest and the canopy closes. With less solar radiation reaching the forest floor, there is better moisture retention and higher humidity near the ground surface.

The effects of these changes are illustrated by the increasingly abundant land snails, where there appears to be a direct relationship between abundance and percent canopy cover. Native forest on Sarigan now supports the largest arboreal snail populations known from the Mariana Islands. For example, *Partula gibba* on Sarigan attains the highest densities recorded for the species, and *Succinea* sp., which may be extinct in the southern islands, may be more abundant than the partulids (CNMI-DFW 2008).

Likewise, *Emoia slevini*, which is the only reptile endemic to the Marianas, has quadrupled in number since the eradication. This species is presently known from five islands in the chain: Alamagan, Asuncion, Guguan, Pagan and Sarigan. Catch rates for this species are now higher on Sarigan than on any other island (CNMI-DFW 2008). Because of this, Sarigan is vital for the survival of this species and could become a source population for future reintroduction efforts.

Bird numbers also appear to be changing. Increased detection frequencies for megapodes are probably linked to increased forage area, especially areas of closed canopy, and an increased prey base in deep forest litter. The reduction in erosion and the addition of leaf litter will further increase forage areas. Similarly, the increase in honeyeaters is probably directly linked to the increased spread of *Erythrina* trees which bloom during a period when other sources of nectar used by the birds are scarce. Since the eradication, abundant *Erythrina* saplings are colonising areas that were once over-grazed grasslands.

Increased cover by native species of plants has been accompanied by increased areas of introduced vegetation. The invasive vine *Operculina ventracosa* is an unplanned consequence of ungulate eradication and had apparently been suppressed by goats. In recent years, the rapid spread of this species has been slowed and may have reached an equilibrium as a result of extended drought during the dry season, intolerance to salt (which can cover the island in the form of salt spray during storms), and competition for sunlight. Two other invasive vines, the mile-a-minute vine (*Mikania micrantha*) and perennial soybean (*Neonotonia wightii*) as well as native trees (including *Erythrina*), all effectively compete with *O. ventracosa* for sunlight.

Anatahan

The use of "Judas" goats with radio transmitters early in the project greatly assisted with locating the remaining animals. "Judas" goats used for the Sarigan project, came from another island and were apparently unable to socialise with the local animals. However, those for Anatahan were local animals released back into their home ranges. These were later readily found from the helicopter and cohort animals dispatched.

Support of the local inhabitants was vital for this project to proceed. There were only a few permanent residents on Anatahan but they all had large extended families on Saipan. These members shared in the resources obtained on Anatahan and held an intention to return to their home island. Shooting the main meat source on the island was thus unpopular and a hard choice for a publicly elected mayor. Discussions with family elders about restoring more culturally desirable natural resources, such as coconut crab (*Birgus latro*) and fruit bat (*Pteropus mariannus*), proved decisive. Anatahan residents understood that crabs and bats need fruiting trees, that goats eat the trees, and that pigs also eat fruit and crabs. Photos showing changes over time helped to convince the residents as did elders' memories of enjoying the shade of forest that had since vanished. In addition, the proposed construction of a fence to contain a sizeable part of the island for goats was acceptable.

After the eruption, the island became uninhabitable and permission was obtained from the residents to remove all ungulates. The eruption not only destroyed the village but apparently also the families' dreams of returning. The residents also accepted that recovery of the island's forest would be more rapid in the absence of ungulates. The CNMI - DFW must now ensure that ungulates are not reintroduced sometime in the future.

Initially, estimated costs for the eradication were about US$2,000,000. So far, the project has cost about US$750,000. After the eruption, operations continued, but funding from the Navy dwindled as the project was delayed due to the eruption, typhoons, governmental bureaucracy and changes to policy. In 2010, the project was in the last year of available funding and only time will tell if pigs will be eradicated. Restrictions on funds and lack of political will have meant that there are no immediate plans for the eradication of other invasive species on the island.

The eruption of Anatahan's volcano seems to have completed the eradication of goats, which have not been observed in four years. The loss of six of the eight radio-collared goats in the initial eruption and the loss of the remaining two in subsequent eruptions support this. Feral pigs were heavily impacted by the eruptions but some large adults (>100 kg) did survive. There were at least four dogs on island before the eradication. After the eruptions, two survived, but are believed to have died out within the year. One cat was shot after the eruptions and sign of more is still being observed. Within the cat's stomach were two rats (*Rattus exulans*) showing that these rodents had also survived. Chickens (*Gallus gallus domesticus*), along with all terrestrial bird species, did not survive. Finally, monitor lizards (*Varanus indicus*) thought to be introduced by ancient Chamorro (Pregill and Steadman 2009), also survived; one was collected in December 2005.

It is estimated that 98 percent of the original forest has been severely altered or totally removed by the eruption. Ground cover was completely buried under at least two meters of ash across the island. Five species of resident land birds were eliminated: Micronesian starling, white-throated ground dove, Micronesian megapode, Micronesian honeyeater, and a unique breeding population of the common buzzard (*Buteo buteo*). The coconut crab (*Birgus latro*) an important resource species is also gone. The Marianas fruit bat, which was one of the largest colonies in the archipelago at about 2000 animals (Worthington *et al.* 2001), was reduced to fewer than ten individuals, but has since increased to about 150 (*pers. obs.*).

Anatahan Island is now practically a "clean slate" and serious thought should be given about developing it into a more desirable pest free environment. There is some interest in continuing bird relocations to Anatahan Island in the future as the forest recovers, in which case the removal of cats and rats should be considered. At present, the most effective method would be an aerial broadcasting of rodenticide with the secondary goal of cat removal. With the current reduced vegetation cover there is a good chance of success. Also without people wanting to return to the island, and the lack of resource species such as the coconut crab and fruit bats which would be a concern, there will be no health or non-target issues. Additionally, a rodent removal operation on Anatahan could be combined with projects on nearby Sarigan and Farallon de Medinilla Islands, with cost savings realized through economy in scale. The removal of rodents and cats from islands in the Marianas would start a new chapter in their recovery and greatly enhance our efforts in protecting and promoting the natural conditions and resources of this unique tropical island system.

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Recent plant eradications on the islands of Maui County, Hawai‘i

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Abstract The state of Hawai‘i (USA) has few regulations to limit plant introductions. A network of interagency island-based invasive species committees has evolved over the past decade to address this vulnerability, with the aim of stopping invasions before they threaten natural areas. On Maui, Moloka‘i, and Lāna‘i, which comprise three of the four islands of Maui County, single-island eradications have been achieved for 12 plant species and eradication is likely imminent for an additional eight species. The islands vary in size, population, and land ownership. We explore the relative importance of those variables in achieving successful eradications along with target species selection, detection strategies, and public support.

Keywords: Invasive plant, *Cortaderia jubata, Cryptostegia grandiflora, Enchylaena tomentosa, Macaranga mappa, Macaranga tanarius, Melastoma sanguineum, Melastoma septemnervium, Parkinsonia aculeata*, *Pennisetum setaceum, Rhodomyrtus tomentosa, Rubus ellipticus*, *Ulex europaeus*

INTRODUCTION

Compared with most locations, species introduced to the Hawaiian Islands establish more readily, can become invasive more rapidly with a shorter lag phase (Daehler 2009; Loope 2011), and often have more severe effects (Denslow 2003). For perspective, Hawai‘i has 50 (Loope 2011) of the "One Hundred of the World's Worst Invasive Alien Species" (Lowe *et al*. 2000) listed by the International Union for the Conservation of Nature (IUCN) Invasive Species Specialist Group (ISSG). Of the 32 invasive species classified as "land plants", Hawai‘i has 20. In this paper, such species are annotated as [IUCN 100].

The Hawai‘i Department of Agriculture can list species as state "noxious weeds", in which case their introduction or transport into uninfested areas is prohibited by state law. This list identifies 79 species, but there have been no updates since 1992 (HDOA 1992). Hawai‘i noxious weeds have been denoted in this text as [HNW].

Given the presence of Hawai‘i Volcanoes and Haleakalā National Parks with their high native biodiversity, and over 300 federally-listed endangered species within the state, especially rigorous efforts might be expected in order to prevent and combat invasions. Such has not been the case, though the amount of effort is probably no worse than that in the USA overall (Loope and Kraus 2009). Conservation management in Hawai‘i has evolved from limited efforts by a few key stakeholders in the 1970s toward substantial and diverse conservation programmes by multiple federal, state and non-governmental agencies. There is also strong support for better measures to prevent and address biological invasions in the age of globalisation (Fox and Loope 2007). However, there are limits on agencies' abilities to adequately address invasive species issues within the state (Kraus and Duffy 2010).

Island-based Invasive Species Committees (ISCs) were formed to help fill identified gaps, starting with an inter-agency group in 1991 on Maui to address the invasion of *Miconia calvescens* [IUCN 100] (Conant *et al.* 1997; Kraus and Duffy 2010). ISCs now cover the six largest Hawaiian islands, with three of those islands in Maui County (Maui, Moloka‘i, and Lāna‘i) served by the Maui Invasive Species Committee (MISC – Maui and Lāna‘i) and the Moloka‘i-Maui Invasive Species Committee (MoMISC).

Statewide efforts are progressing to institute weed risk assessments (Daehler *et al.* 2004), prevent sanctioned planting of pest plants, and stop new invasive introductions to individual islands and the state despite the limited regulation of plant introductions. The ISCs are poised to address this vulnerability, with the principal aim of stopping invasions before they threaten natural areas.

In Maui County, challenges posed by invasive species include protecting about 120 federally endangered plant species from weed and pest incursions, plus operating across three different islands, a diversity of habitats, and a range of socio-economic conditions. Habitats in the county stretch from sea level to >3000 m, in rainfall zones from very wet (annual rainfall > 8000 mm) to very dry (annual rainfall < 300 mm), including coastal shrub, dryland, mesic, and rain forest and alpine vegetation zones (Ziegler 2002). Many species of invasive plants already occupy a wide range of climatic zones on the islands and pose immediate or eventual threats to endemic species of plants, animals, and natural areas.

People are a crucial component of invasive species management programmes (García-Llorente *et al.* 2008). Introductions of exotic species are likely to increase with island area, population size, economic activity, and accessibility to air travel (Denslow *et al.* 2009; Kueffer *et al.* 2010). Introduction rates affect whether targeted species can be detected in all locations and the potential for reintroduction. Thus, information about the physical and socioeconomic conditions of Maui County may be relevant to evaluating overall success.

Our paper describes progress with advanced efforts to eradicate 12 plant species, with an additional eight species on target for eventual eradication. We consider programmatic and socio-economic factors associated with successful eradications. For purposes of this paper, eradication means: removal of all known individual plants for a given species from all known locations. For some species, the eradication process includes ongoing visits to address recruitment from known seedbanks.

MATERIALS AND METHODS

Study area

The islands of Maui County are linked politically but vary in size, population growth, and extent of private and publicly owned lands (Table 1). Maui is the largest and most populated island. From 2000-2010, Maui experienced a 13% growth rate, in contrast to Moloka‘i at 1%, and Lāna‘i, which slightly decreased. The islands vary in accessibility. Maui is served by direct flights from

Pages 325-331 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

Table 1 Island size, population growth, and land tenure.

Island	Size[1] (km²)	Population[2] (2010)	Population 2000-2010 % Change[3]	Ownership: Private/Public[4] %
Maui	1884	144,444	+12.8	65/35
Moloka‘i	674	7345	+1.2	70/30
Lāna‘i	364	3135	-1.8	99/1

[1]Juvik and Juvik 1998.
[2]U.S. Census Bureau, 2010 Redistricting Data (Public Law 94-171) Summary File P1.
[3]U.S. Census Bureau, Census 2000 Summary File 1: Hawaii.
[4]Based on Maui County GIS tax map information.

the mainland and all other Hawaiian islands. Flights to Moloka‘i are available only from Maui and O‘ahu, while Lāna‘i is accessible by regular commercial airlines only from O‘ahu. In 2009, Maui had nearly 1.9 million arriving air passengers, compared to approximately 48,000 for Moloka‘i, and 61,000 for Lāna‘i. (Maui County Data Book 2010). Maui has more than 60 plant providers or landscapers, many of which import plants from the island of Hawai‘i and the U.S. mainland. Moloka‘i has no major plant supplier and Lāna‘i has a single nursery that provides plants to two resort areas on the island. Land ownership varies by island. The highest percentage of private land ownership (ca 99%) is on Lāna‘i.

Target selection

There are two ways for an invasive plant species to become targeted for eradication in Maui County: review during an annual priority-setting process held by each ISC, or as a rapid response to a newly-discovered species brought to the committee's attention at a regular (bi-monthly) meeting. New discoveries of incipient species are typically made by committee members, staff, or other resource professionals in the community. With the exception of several species targeted for containment, such as *Miconia calvescens* or *Cortaderia jubata* on Maui, the objective for any new plant species is eradication.

Evaluation criteria include: risk to the island's environment, health, agriculture or economy, with special emphasis on environmental threats, feasibility, and cost of management options. Information about the relative risk posed by a potential target derives from several sources, including the expert knowledge of committee members and other local botanists, use of the Hawai‘i Pacific Weed Risk Assessment (HPWRA) (Daehler *et al.* 2004; www.hpwra.org), and literature review, including Internet searches and general references such as Randall (2007) and Weber (2003). For early eradication targets, the HPWRA tool was not available during initial feasibility analyses. Eradication feasibility considers biological factors such as seed dispersal mechanisms and seed longevity, and extent of infestation. Many of the species reported herein were identified as potential eradication candidates as the result of a roadside survey and expert interviews conducted in 2000 (Starr *et al.* 2011).

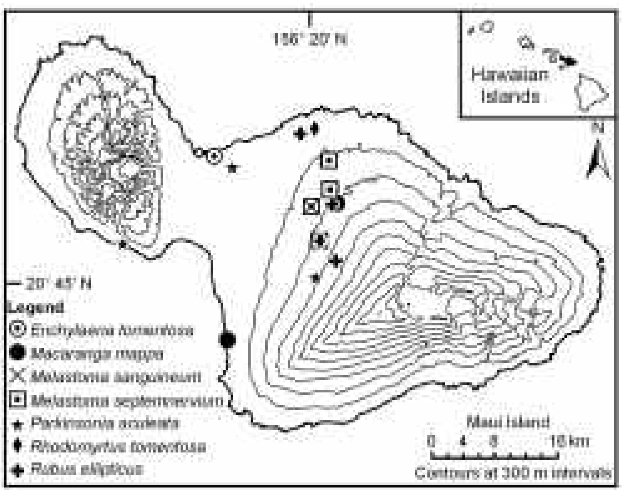

Fig. 1 Plant eradications on Maui, Hawai‘i.

Survey and Management Techniques

Any eradication campaign against plants must adequately address three components: delimitation or determining the known extent of the invasion (Panetta and Lawes 2005), containment (no evidence of spread), and extirpation (Panetta 2007). Delimitation methods included active and passive strategies (Dewey and Anderson 2004), which involved roadside surveys, backyard searches in residential areas, and ground sweeps in rural or wildland areas. These were all conducted by a trained field crew at the initial detection site and surrounding areas. Roadside surveys on Maui were conducted in 2000 and 2009 (Starr *et al.* 2011) by two botanists driving all paved roads searching for a list of specific plants, including those covered in this paper.

Facilitation of passive surveys focused on teaching the public how to identify target species. Activities included 19 early detection workshops since 2008 for conservation workers; field professionals such as county road workers and parks and recreational staff, dock workers, federal agricultural inspectors; and members of the general public. Participants received an informative field guide about the target species (http://pbin.nbii.org/reportapest/maui/mauiearlydetectionguide_2008052.pdf). Publication of articles in the local newspaper highlighted early detection targets. (http://www.hear.org/misc/mauinews/). Outreach professionals attended community events and worked with local schools to inform the public about target species. The U.S. Geological Survey's Pacific Basin Information Node spearheaded a multi-agency reporting system to facilitate rapid response to incipient pests, which includes an online reporting tool (www.reportapest.org). These activities have resulted in valid reports from the public.

Management work at each infested site was conducted by ISC staff or partner agencies. Work on private lands was performed after obtaining permission from the landowner. Eradication techniques included hand-pulling or treatment with herbicide. Seed heads from flowering grasses were typically cut and bagged before treatment with herbicide. Geospatial information was collected at each infested site. More specific information about eradication techniques and plant locations is available on request. Eradications

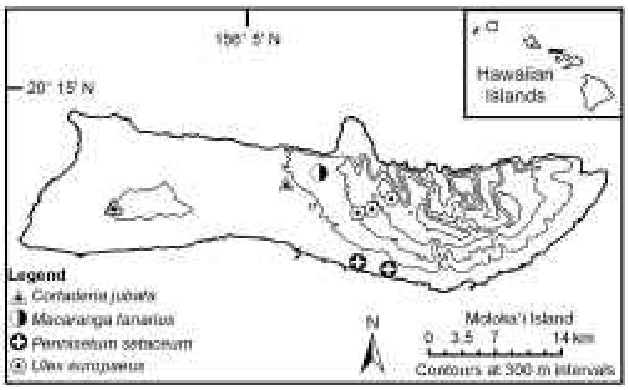

Fig. 2 Plant eradications on Moloka‘i, Hawai‘i.

Table 2 Number of plants removed by year, 2001-2009.

Island	Species	2001	2002	2003	2004	2005	2006	2007	2008	2009
Moloka‘i	*Cortaderia jubata*	5	-	-	-	-	-	-	-	1
Lāna‘i	*Cryptostegia grandiflora*	-	-	-	-	-	3	-	-	-
Maui	*Enchylaena tomentosa*	4	10	-	-	-	-	-	-	-
Maui/Lāna‘i	*Macaranga mappa*	-	-	-	3	-	1	2	6	1
Moloka‘i	*Macaranga tanarius*	-	-	-	-	-	-	1	-	-
Maui	*Melastoma sanguineum*	-	-	-	1	-	-	-	-	-
Maui	*Melastoma septemnervium*	-	1	-	-	-	1	-	-	-
Maui	*Parkinsonia aculeata*	17	-	-	-	-	-	-	-	-
Moloka‘i	*Pennisetum setaceum*	-	-	4	2	-	-	-	-	-
Maui	*Rhodomyrtus tomentosa*	-	152	12	1	-	-	-	-	-
Maui	*Rubus ellipticus*	1	1	-	-	-	-	-	-	1
Moloka‘i	*Ulex europaeus*	-	24	36	17	8	2	-	-	-

were achieved by repeat visits to known infested sites at intervals designed to ensure that plants did not fruit or set seed. Information about seed longevity was considered in determining the likelihood that a remaining seedbank had been exhausted. Site visits and surveys of surrounding areas continue to be made around all known locations of target species.

RESULTS

Seven plant species were eradicated from Maui: *Enchylaena tomentosa, Macaranga mappa, Melastoma septemnervium, Melastoma sanguineum, Parkinsonia aculeata, Rhodomyrtus tomentosa,* and *Rubus ellipticus* (Fig. 1, Table 2). Four species were eradicated from Moloka‘i: *Cortaderia jubata*, *Macaranga tanarius*, *Pennisetum setaceum*, and *Ulex europaeus* (Fig. 2, Table 2). Two species were eradicated from Lāna‘i: *Cryptostegia grandiflora* and *Macaranga mappa* (Fig. 3, Table 2).

Two species were on the IUCN list of 100 Worst Invaders and approximately half (7) were Hawai‘i noxious weeds; all but one subsequently scored as “High” risk under the HPWRA (Table 3). None of the species was present on more than three sites on any island. The largest number of plants killed was 165 plants of *R. tomentosa*. Excluding

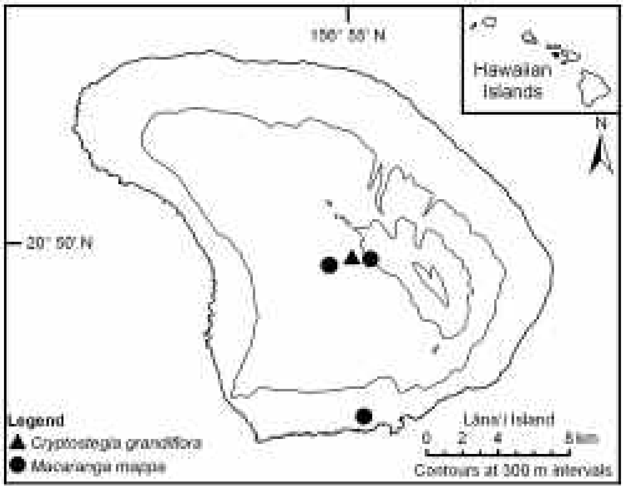

Fig. 3 Plant eradications on Lãna‘i, Hawai‘i.

Table 3 Characteristics of plant species eradicated in Maui County.

Species	# of Sites[1]	# of Plants	Effort (hrs.)	Area[2] (ha.)	State Noxious Weed[3]	HPWRA Rating[4]	Land Tenure	Seed Longevity[4]
Cortaderia jubata	3	6	16	11	X	High	Private	< 1 yr
Cryptostegia grandiflora	1	3	1	0.3		High	Private	1-5 yrs
Enchylaena tomentosa	1	14	14	0.4		Low	Public	> 1 yr.
Macaranga mappa[5]	3/3	7/6	15/13	3/8.2		High	Private	> 1 yr
Macaranga tanarius	1	1	2	0.4		High	Roadside	Unknown
Melastoma sanguineum	1	1	2	0.1	X	High	Private	Unknown
Melastoma septemnervium	2	2	8	1.1	X	High	Private	Unknown
Parkinsonia aculeata	3	17	8	0.3		High	Roadside	> 1 yr.
Pennisetum setaceum	3	6	33	1.9	X	High	Private	6 yrs.
Rhodomyrtus tomentosa	2	165	91	4.5	X	High	Private	> 1 yr
Rubus ellipticus	3	3	1	1.6	X	High	Private	> 1 year
Ulex europaeus	2	87	47	2.2	X	High	Prvt/Public	> 30 yrs

[1]A site is defined by property ownership.
[2]Area (hectares) is the infested area or area surveyed.
[3]Listed as a noxious weed by the Hawai‘i Department of Agriculture, Hawai‘i Administrative Rules § 4-68.
[4]See www.hpwra.org for Risk Assessments & references for seed longevity.
[5]Data for Maui/Lāna‘i.

roadside surveys, no area surveyed was >11 hectares. Most eradications were on private land, the only exception being *E. tomentosa*, which was solely on public land. The second roadside survey (Starr *et al.* 2011) was conducted nine years after the first and helped boost our confidence that the infestations had not spread beyond known areas; no new locations of the eradication targets were discovered during the 2009 surveys.

The following outlines the justification for each of the target species discussed in this paper and highlights eradication efforts.

Cortaderia jubata **(Lem.) Stapf. – Poaceae [HNW]**

Maui has two *Cortaderia* spp., *C. jubata* (jubata grass, pampas grass) and *C. selloana* (pampas grass). Both species are ornamental bunch grasses capable of long distance wind dispersal and are known as aggressive weeds in numerous locations (Weber 2003). So far, only *C. jubata* is highly invasive on Maui (Loope 1992), but in California (the most likely genetic source of both species for Hawai'i), *C. selloana* is equally if not more invasive and damaging (Lambrinos 2001). *Cortaderia jubata* is native to Bolivia, Ecuador and Peru; horticultural stock apparently consists of a single genotype and is from southern Ecuador (Okada *et al.* 2009).

In Hawai'i, *C. jubata* was introduced for ornamental planting and was discovered invading natural areas on Maui in 1989. This species has established in numerous areas of rain forest as well as bogs on East and West Maui and has been detected and controlled in Haleakalā National Park. With *C. selloana*, *C. jubata* comprises the second highest plant priority for MISC; management efforts span thousands of hectares and involve ground work in residential and wildland areas and aerial operations in more remote areas. The limited distribution of *C. jubata* on Moloka'i made it a strong candidate for eradication. The species was first discovered at two sites in 2001 and considered eradicated after seven years of monitoring, when another site was detected during island-wide surveys for the *Babuvirus* (banana bunchy top virus) [IUCN 100]. The landowner had purchased seeds over the Internet. The homeowner was given a native plant as a replacement, and *C. jubata* at the new site was removed in 2009 before it set seed.

Cryptostegia spp. **- Asclepiadaceae**

Cryptostegia (rubber vine) is a genus endemic to Madagascar. There are two species, *Cryptostegia grandiflora* R. Br. and *C. madagascariensis* Bojer ex Decne.(GRIN n.d.), both of which are usually identified as *C. grandiflora*. Careful inspection has revealed that nearly all Hawaiian cultivated plants are *C. madagascariensis* (Staples *et al.* 2006). Both species have been spread by the plant trade, have become invasive in far-flung locations of the world, and have sap toxic to livestock. In Australia, *C. grandiflora* is a "Weed of National Significance" notorious for invasion of 40,000 km^2 in the Australian wet tropics, where it covers whole forests (Tomley and Evans 2004). *Cryptostegia madagascariensis* has recently been discovered invading unique riverine forests of northeastern Brazil (da Silva *et al.* 2008). Biological control exploration and testing has been underway for agents for *C. grandiflora* since 1985 in Australia and is now underway for *C. madagascariensis* in Brazil (da Silva *et al.* 2008).

Cryptostegia was recorded as naturalised on several of the main Hawaiian islands, including Moloka'i (Staples *et al.* 2006) and O'ahu (Frohlich and Lau 2008). On Lāna'i, the species was detected at a single residential location in Lāna'i City during 2006. The cooperative landowner had removed the plants by mid-2007. No recruitment has been observed at the site and it is considered eradicated from Lāna'i. On Maui, *Cryptostegia* is on several residential properties where it has been planted as an ornamental, but eradication remains elusive owing to landowner recalcitrance. These sites are potential sources of further invasion via the readily wind-dispersed seeds. On Moloka'i, *C. madagascariensis* Bojer has been the subject of an aggressive eradication campaign, and although root suckers remain, the species is considered en route to eradication.

Enchylaena tomentosa **R. Br - Chenopodiaceae**

Otherwise known as ruby or barrier saltbush, this small shrub is native to Australia, and had been reported as naturalised in New Caledonia (Imada *et al.* 2000) and Israel (Danin 2000). The invasiveness of *E. tomentosa* in New Caledonia is perhaps questionable, since it is not cited by local botanists (J. Munzinger, Herbarium IRD pers. comm.). Development of the Weed Risk Assessment tool occurred in Hawai'i after control of this species and subsequently ranked it as a "Low" risk. In Hawai'i, *E. tomentosa* was known only from one location on Maui (Imada *et al.* 2000), within Kanahā Pond, a state coastal wildlife sanctuary in Central Maui. Removal of four *E. tomentosa* plants was considered an early (2001-2002) success. An additional 10 plants were removed and no additional plants have been detected at this site, which is regularly surveyed by state wildlife personnel (F. Duvall, Hawai'i Department of Land and Natural Resources, Maui pers. comm.).

Macaranga mappa **(L.) Müll. Arg. - Euphorbiaceae**

Commonly called bingabing, this species is native to the Malesian biogeographic region of Malaysia but has naturalised in Hawai'i on the islands of O'ahu and Hawai'i (Wagner *et al.* 1999). Its abundance in some areas is attributed to forestry plantings in the late 1920s (Skolmen 1980). The species has spread into the forested areas of the eastern coast of Hawai'i, where stands of the large-leaved *M. mappa* create deep shade, its dense growth habit crowds out other vegetation, and it demonstrates strong regeneration capacity associated with its large seed bank (Cordell *et al.* 2009).

Macaranga mappa and *M. tanarius* are easily identified by their large umbrella-like leaves. A single large *M. mappa* tree in upcountry Maui was found during roadside surveys (Starr *et al.* 2011). This intentional planting was removed in 2004. The species was subsequently detected and removed at two additional locations on Maui, with no apparent connection to the initial site (Fig. 1). *Macaranga mappa* was detected on Lāna'i at three sites in small numbers in 2007 and 2008. Two of the three Lāna'i *M. mappa* sites were apparently the result of contaminated soil or nursery stock from the island of Hawai'i and this was likely the case for the two other sites on Maui.

Macaranga tanarius **(L.) Müll. Arg. - Euphorbiaceae**

This parasol leaf tree is native to Southeast Asia, Papua New Guinea, and Australia (GRIN n.d.). Similar to *M. mappa*, the species was an intentional forestry introduction in the 1920s, now forms dense thickets where it has become established, and is naturalised on O'ahu and Kaua'i (Wagner *et al.* 1999). Extensive infestations are also in the valleys and disturbed areas of West Maui; on East Maui it has been the target of localised removal. *Macaranga tanarius* was detected in a single location on Moloka'i and removed in 2007; no recruitment was ever observed at the site.

Melastoma spp. **– Melastomataceae [HNW]**

Hawaiʻi has two invasive *Melastoma* species. *Melastoma septemnervium* Lour. (Asian melastome) is native to southern China, Vietnam, Taiwan, the Ryukyu Islands and southern Japan; and *M. sanguineum* Sims (red melastome) is native to the Malay Peninsula, Java, Sumatra, Vietnam, and southeastern China (Staples and Herbst 2005). The two similar species have been recognised as serious pest plants in Hawaiʻi since about 1960 (Plucknett and Stone 1961). The entire *Melastoma* genus has state noxious weed status. Both species were grown as ornamentals for their showy flowers, shrubby habit, and attractive foliage. However, they outcompete native plants by forming dense monospecific thickets, growing up to 2 m tall, at elevations up to 900 m. Extensive infestations of *M. septemnervium* (aka *M. malabathricum,* but name misapplied) are now found on Kauaʻi and Hawaiʻi, and a relatively recent infestation was found on Oʻahu (as *M. candidum*, a synonym, Conant 1996). *M. sanguineum* is naturalised only on the island of Hawaiʻi. *Melastoma septemnervium* was found and removed at two sites on Maui, where it was last detected in 2006. *Melastoma sanguineum* was removed from one site on Maui in 2004.

Parkinsonia aculeata **L. - Fabaceae**

Commonly called Jerusalem thorn, this species is native to South America and the West Indies (Staples and Herbst 2005) and probably other sites in Southwestern North America. It is most notoriously invasive in Australia, where it forms dense, thorny, impenetrable thickets, with seeds dispersing along rivers, streams, and gulches; it is one of about 20 Australian "Weeds of National Significance" (www.weeds.org.au/WoNS/) and is a biological control target in Australia (van Klinken 2006). On Oʻahu, the species was introduced by the U.S. Army (Staples and Herbst 2005), but eradication was requested by the Hawaiʻi Territorial Board of Agriculture before 1920. *Parkinsonia* is likely to be sparingly naturalised on Kauaʻi and Oʻahu. On Maui, *P. aculeata* was detected at two locations, with one of those removed voluntarily by the landowner. A single planting on West Maui was also targeted for eradication. All plants were removed by 2001.

Pennisetum setaceum **(Forssk.) Chiov. – Poaceae [HNW]**

Commonly known as fountain grass, this aggressive early coloniser of lava fields and dry forests covers tens of thousands of hectares on the island of Hawaiʻi. It destroys native communities by increasing fire frequency and limiting germination, survival, and growth of native dry forest species (Williams *et al.* 1995; Cabin *et al.* 2002). The native range of *P. setaceum* spans much of the Middle East and North Africa, but is primarily arid coastal regions of the Sahara Desert (Williams *et al.* 1995; Le Roux *et al.* 2007). The species thrives from sea level to 2800 m elevation in Hawaiʻi despite lack of genetic variation (Williams *et al.* 1995), and is now on all the main Hawaiian islands (Wagner *et al.* 1999; Starr *et al*. 2011). On Maui, *P. setaceum* has been targeted for eradication since about 1976 (Loope 1992), with the successful exhaustion of seedbanks from nine known small populations. A much larger infestation exists on Lānaʻi, where the species is targeted for containment. *Pennisetum setaceum* was known from only two sites on Molokaʻi, detected at two different times. One involved contaminants from bird seed and the other site had plants brought to Molokaʻi from the island of Hawaiʻi. Removal on Molokaʻi was completed by 2004, with no subsequent detection.

Rhodomyrtus tomentosa **(Aiton) Hassk. – Myrtaceae [HNW]**

This downy rose myrtle, which is native to Southeast Asia, is established and invasive on Kauaʻi, Oʻahu and Hawaiʻi (Wagner *et al.* 1999). The evergreen shrub is fire-adapted, can tolerate a wide range of environmental conditions, and is highly invasive in Florida (Langeland and Burks 1998) and on the island of Raiatea in French Polynesia (Meyer 2004). On Kauaʻi, *R. tomentosa* blankets portions of the lower-elevation landscape, covering thousands of hectares (Burney and Burney 2007). The species was detected on Maui at two locations; eradication efforts, which began in 2002, had concluded by 2004.

Rubus ellipticus **Sm. - Rosaceae [IUCN 100] [HNW]**

Commonly called yellow Himalayan raspberry, this thorny thicket-forming shrub has long (to 4 m) trailing shoots, is native to areas of temperate and subtropical Asia, exhibits aggressive growth, and is difficult to control. In Hawaiʻi, *R. ellipticus* has become well established in the Hawaiʻi Volcanoes National Park and surrounding areas, where it threatens native resources (Stratton 1996). The species has been transported to Maui as a contaminant in mulch or tree fern trunks (*Cibotium* spp.), which are sold and shipped from the island of Hawaiʻi.

Rubus ellipticus was first discovered on Maui in 1997. Eradication efforts over the next five years by MISC partners ensured that plants never fruited (S. Anderson, Haleakalā National Park, Maui pers. comm.). There have been two discoveries since then, in each instance the result of contaminated plants shipped between islands. One site is a botanical garden, where there was repeated regeneration from stock deep in the trunk of a tree fern, despite the owner's attempts to eradicate it (F. Starr, University of Hawaiʻi, Maui pers. comm.). Efforts to manage *R. ellipticus* must realistically be regarded as a "serial eradication" as long as unregulated interisland transport of plants continues.

Ulex europaeus **L. – Fabaceae [IUCN 100] [HNW]**

Widely known as gorse, this notorious woody shrub is native to Britain and parts of Europe, forms impenetrable thickets, excludes grazing animals, and makes land unusable where it persists. *Ulex europaeus* has extensively invaded pasturelands and native ecosystems on Hawaiʻi and Maui; substantial biocontrol efforts in Hawaiʻi to date have not been effective (Markin *et al.* 2002). Gorse's long seed viability, reported as 50 years or more, make this a challenging target for eradication (Motooka *et al.* 2003). The discovery of low numbers on Molokaʻi (Conant 1996) at three locations, including a forested area, suggested eradication was still feasible. No plants have ever been detected outside the treatment area and none observed since 2006.

Other species

In addition to the successes outlined above, eight more invasive plant species are on target for eventual eradication within Maui County: on Maui these are *Acacia retinodes*, *Maclura pomifera*, *Silybum marianum*, and *Verbascum thapsus*; on Molokaʻi these are *Arundo donax* [IUCN 100], *Cryptostegia madagascariensis*, *Salsola kali*, and *Setaria palmifolia*. The known extents of these populations have been delimited and efforts are focused on exhausting seedbanks or controlling sprouts from vegetative re-growth.

DISCUSSION

Efforts to eradicate 12 species in Maui County have been relatively successful and were accomplished at low cost, consistent with the concept that early detection and rapid response are cost-effective means of addressing invasive species. All but one species targeted for eradication were known to be highly invasive plants.

Key factors for successful eradications included: appropriate target selection, including low numbers of plants on few properties; persistent efforts by trained crews; and cooperative landowners. Most eradications were completed within a one- to two-year time frame, with the longest effort extending over five years for *Ulex europaeus*. Seedbanks exist for some species; thus, continued vigilance is essential, although only *U. europaeus* has a particularly persistent propagule bank. Precise geospatial information along with the institutional memory of key staff and partners boost our confidence that seedbanks can eventually be exhausted for all target species. Continued financial support from local, state, and federal agencies will be necessary to ensure repeat site visits.

Maui's larger population and higher rate of population growth, its enhanced air accessibility, and more horticultural businesses, mean more opportunities for weedy plants to be introduced to the island. Its larger overall size and more private properties also complicate detection efforts. In contrast, field staff are able to regularly survey the single nursery on Lāna'i for target species and visit almost every property on the island during annual surveys. Moloka'i has not had a commercial nursery in recent years and its smaller community makes it possible to reach most residents during major outreach events. Thus, the level of confidence associated with eradications on Maui must be considered lower than those for Moloka'i and Lāna'i. The possibility of reintroduction exists on all islands, as demonstrated by the Internet purchase of *Cortaderia jubata* seeds on Moloka'i and reinvasion of *Macaranga mappa* and *Rubus ellipticus* as contaminants in nursery stock from the island of Hawai'i.

In the absence of meaningful regulations mandating removal of invasive species, eradications can only be achieved through landowner cooperation. All but two of the eradications were achieved on private lands, underscoring the importance of strong public support. On Moloka'i, initial resistance to control of *Cortaderia jubata* was overcome. Eradication efforts on Lāna'i were facilitated by strong cooperation from Lāna'i residents, the majority of whom live in the island's main town, and access to open areas by the primary landowner. Eradication remains elusive for *Cryptostegia* and *Acacia podalyriifolia* on Maui because landowners are refusing to cooperate. Landowner recalcitrance is also thwarting efforts to control the more entrenched *C. jubata* and *C. selloana* on Maui, even though *C. jubata* is a state noxious weed.

While these eradications are viewed as successes, they do not constitute the major focus of work, at least on Maui. Compared to MISC's work on all invasive species, resources devoted to the reported eradications represented approximately 1% of total personnel effort over the period of the project. In contrast, over $1 million is currently being spent annually to contain *Miconia calvescens* and *Cortaderia* spp. on Maui. Smith (2002) articulated Hawai'i's need to accelerate efforts at biological control for some of the most damaging invasive plant species to avoid obliteration of large expanses of native ecosystems; the need remains. *Miconia calvescens* is by far the greatest threat to biodiversity and endangered plant species, but other ominous threats include the shrub–tree strawberry guava (*Psidium cattleianum* [IUCN 100] Myrtaceae); the large herb kahili ginger (*Hedychium gardnerianum* [IUCN 100] Zingiberaceae); the shrub *Clidemia hirta* [IUCN 100], another member of the Melastomataceae; and several other serious weeds (Stone *et al.* 1992). For certain widespread, high-impact weeds, biological control is an essential part of the mix needed for conservation of the biodiversity in Hawai'i – given that there appears to be no other conceivable long-term solution. Despite this urgency to expand biocontrol efforts, the current focus on measures to exclude potential new invasive species and eradicate incipient invasives is a continuing high priority (Kraus and Duffy 2010).

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Introduced mammal eradications in the Falkland Islands and South Georgia

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Abstract Within the past decade, Norway rats (*Rattus norvegicus*) have been removed from 20 islands in the Falkland Islands and one island at South Georgia. The islands range in size from <1 to 305 hectares. Islands were selected on the basis of accessibility, size, distance offshore, operating cost, landowner support, and availability of suitable habitat for threatened bird species. The assumption that these islands have a high potential for re-colonisation by native species has been confirmed by the return of tussacbirds (*Cinclodes antarcticus*) and South Georgia pipits (*Anthus antarcticus*) and an apparent increase in the size of white-chinned petrel (*Procellaria aequinoctialis*) and sooty shearwater (*Puffinus griseus*) populations. With no helicopters available, the main method of bait application was hand broadcasting. Initially, local operators were supervised by New Zealand experts, thereby gaining the experience required to run their own programmes which now employ local fieldworkers. Campaigns between 2000 and 2009 used Pestoff 20R (20 ppm brodifacoum) cereal-based pellets and Ditrac wax blocks (50 ppm diphacinone). Recent developments in the Falklands include the first multispecies eradication attempt. Patagonian grey foxes (*Lycalopex griseus*) were eradicated from 320ha Tea Island in 2008, followed by Norway rats in 2009.

Keywords: Norway rats, *Rattus norvegicus,* Patagonian grey fox, *Lycalopex griseus*, hand broadcasting, trapping

INTRODUCTION

The Falkland Islands and South Georgia are located in the South Atlantic Ocean between 51°S and 54°S. The Falklands are farmed and inhabited by 3000 people who privately own approximately 75% of all land. South Georgia is Crown land with no permanent human population. Both island groups have an exceptional abundance of seabirds and no native terrestrial mammals.

Bird populations in both island groups have been significantly impacted by introduced predators, beginning in the late 1700s when Norway rats (*Rattus norvegicus*), ship rats (*R. rattus*) and house mice (*Mus musculus*) arrived on whaling and sealing vessels. Cats (*Felis catus*), feral pigs (*Sus scrofa*) and Patagonian grey foxes (*Lycalopex griseus*) were introduced to the Falklands with significant impacts on native birds (Strange 1992; Woods and Woods 1997). Species that are particularly vulnerable to rat predation in the Falklands are the tussacbird (*Cinclodes antarcticus*), Cobb's wren (*Troglodytes cobbi*) and several species of burrowing petrels. At South Georgia, populations of the endemic South Georgia pipit (*Anthus antarcticus*) and burrowing petrels have been heavily reduced by Norway rats (Pye and Bonner 1980; McIntosh and Walton 2000).

The Falkland Islands (12,200 km^2) lie 450 km north-east of Tierra del Fuego and north of the Polar Front. The archipelago encompasses about 260 km by 140 km with a maximum elevation of 705 m. The climate is temperate oceanic, with an average annual temperature at sea level of 6°C, ranging from 2°C in the winter to 10°C in summer. Annual precipitation varies from 300 mm at the west end of the group to 600 mm at the east; average wind speed is about 16 knots. The main vegetation type is oceanic heathland dominated by whitegrass (*Cortaderia pilosa*) and diddle-dee (*Empetrum rubrum*) with remnant stands of tussac (*Poa flabellata*) now mostly restricted to ungrazed offshore islands. The Falkland Islands Biodiversity Database held by the Falklands Islands Government lists over 700 islands in the archipelago. Excluding the two main islands of East and West Falklands, islands range in size from 21,800 ha to small stacks. About 600 islands are covered in oceanic heath or tussac. Of these, more than 400 are known to have exotic terrestrial mammalian predators, at least 130 are believed to be rat-free, and the rest are unsurveyed.

South Georgia (3755 km^2) lies south of the Polar Front and approximately 1450 km east-south-east of the Falklands. The island is 170 km long, between 2 and 40 km wide, and rises to 2960 m. Mean temperature at sea level is -1.2°C in winter when snow covers most of the island, rising to 5°C in the summer. More than 50% of the island is under permanent ice with many large glaciers reaching the sea. Virtually all flora and fauna are found along the coastal margins. The dominant vegetation is tussac grassland. Tussac also provides Norway rats and house mice with food and shelter, and is the key to their survival at this latitude and extreme climate. Two thirds of South Georgia's 1300 km mainland coastline is inhabited by rats, and a further 50 km are known to have house mice. Rats are also recorded on at least 50 offshore tussac islands, including Saddle Island (103 ha) which has been colonised within the past 20 years. This island is separated from the mainland by a 270 m wide passage and was last recorded as rat-free in 1987 (S. Poncet data).

Successful eradications of Patagonian grey foxes and feral cats in the Falklands were carried out by farmers at least as early as the 1960s, but only in the last decade has the knowledge, funding and public support become available for rat eradication campaigns. Between 2000 and 2009, eradication of Norway rats has been attempted on 39 islands in the Falklands and one island (Grass Island) at South Georgia. Islands range in size from less than 1 ha to 320 ha, where an island is defined as land that is completely surrounded by water at lowest astronomical tide.

Organisations running invasive mammal eradication programmes in the Falklands are the conservation interest group Beaver Island LandCare (BILC) and the charity Falklands Conservation (FC). Funding sources include the United Kingdom's Foreign and Commonwealth Overseas Territories Environmental Programme, Falkland Islands Government (FIG), the RSPB's South Atlantic Invasive Species Programme, Falklands Conservation and the Antarctic Research Trust.

In 2001, FC commissioned Derek Brown, Lindsay Chadderton and Kerry Brown from New Zealand to undertake a series of Norway rat eradications with FC staff and volunteers. The New Zealanders also drafted

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"Guidelines for Eradication of Rats from Islands within the Falklands Group", developed criteria for prioritising islands selected for rat eradications and proposed an island restoration plan (Brown 2001). At South Georgia, rat eradication plans for the entire island are being prepared by the South Georgia Heritage Trust.

METHODS

In the Falklands, islands were usually selected for eradication on the basis of landowner support, terrain accessibility, size, distance offshore, operating cost and habitat suitability for re-establishment of threatened bird species. The targeted species were the Norway rat and Patagonian grey fox. Rat eradication operations used bait stations on two islands, and hand broadcasting on the remainder. Leghold traps and snares were used to remove foxes. To date, there have been no attempts to eradicate house mice or ship rats.

Rat eradication by hand broadcasting

There are no commercial helicopters available in the Falklands, so rat eradication has been achieved principally by hand broadcasting of either Pestoff 20R 2 g cereal-based pellets (active ingredient 20 ppm brodifacoum) or Ditrac 28 g wax blocks (active ingredient 50 ppm diphacinone). Operations are scheduled towards the end of winter (August/September) when rat numbers are lowest and food is scarce. With the return of burrow-nesting Magellanic penguins (*Spheniscus magellanicus*) in mid-September, food for rats, such as guano and regurgitations, becomes increasingly abundant.

The hand broadcasting method for Ditrac blocks was developed by BILC between 2007 and 2009 on 11 islands in the Beaver Island group. It was designed to replicate an aerial baiting operation, following recommendations from New Zealand experts Andy Cox and Ian McFadden of the Department of Conservation (DOC), and Derek Brown who have advised on, and participated in, eradications in South Georgia and the Falklands since 2000.

Each operation consisted of the following stages.

1. Surveys of the terrain, wildlife and habitat at the target eradication islands and also of islands and mainland areas in the vicinity each island, in order to assess: a) rodent status, habitat types, bird abundance and distribution and suitable habitat for re-colonisation by tussacbirds, Cobb's wrens and burrowing petrels; b) re-invasion potential from adjacent islands or mainland areas; and c) the feasibility and logistical requirements of an eradication operation.

2. Submission of an Operational Plan to the Falkland Islands Government's Environmental Planning Department and the land owner for review.

The plan included designs of the baiting grid using mapping software OziExplorer for a bait spread regime of 4 kg/ha on inland areas and 8 kg/ha on the coast and in dense vegetation such as tussac.

For the two largest islands treated (Tea Island 320ha and Governor Island 270 ha), tracks were created for a central 'backbone' line down the middle of each island. This central line was the starting point for cross-island transects that were 50 m apart and ran at right angles from either side of the central line out to the coast. On the smaller islands, cross-island transects started from the coast and headed parallel across to the opposite coast. Each transect line was individually numbered. Co-ordinates (waypoints) were also created for the position of bait depot points along the transect lines. These depot points were flagged by bamboo canes. The distance interval between depot points along each line was 200 m for a baiting regime of 8 kg/ha and 400 m for 4 kg/ha. A map displaying the pre-established numbered transect lines and depot points was given to each operator.

The depot points were positioned using hand-held GPS units uploaded with the pre-determined waypoints and tracks. Each depot point was individually numbered.

3. On site, one bait tub (a sealed plastic bucket containing 8 kg of bait) was deposited at each depot point. The number of the depot point was written on each tub. The bait was hand broadcast by 2 to 6 operators, depending on the size of the island and operator experience. Operators walked as a front, one along each cross-island transect line, using hand-held GPS units to follow GPS tracks while broadcasting bait. Any gaps in coverage were detected by the units which recorded tracks walked while broadcasting. Each operator collected a tub at each depot point and spread its contents along the interval between points. For a baiting regime of 8 kg/ha, 14 blocks of bait were broadcast every 10 m (7 blocks were broadcast every 10 m for 4 kg/ha). The broadcast swathe was approximately 30 m, with 5 blocks thrown to the left, 5 to the right and 4 at the feet of the operator. Along the coastline, one operator distributed one tub (8 kg) of bait every 100 m. Once baiting was complete, all equipment was removed from the island.

4. Submission of a post-baiting report to FIG's Environmental Planning Department.

5. Post-baiting checks were conducted at the end of the second summer after baiting to search for fresh rat sign and check chew sticks (edible oil-soaked pine sticks) deployed three months or longer after baiting.

Fox eradication by trapping

The 2008 BILC fox eradication programme on Tea Island adopted the Alaska Maritime Wildlife Refuge's methods for fox trapping in the Aleutian Islands (Ebbert 2000). Steve Ebbert of the US Fish and Wildlife Service visited the Falklands in March 2008 to advise on the campaign. Four local operators were trained by Rick Ellis, a trainer-trapper from Alaska who also supervised the first phase of the Tea Island operation that ran from 15 September to 25 October 2008. Sets included 8 snares and up to 80 leghold traps baited with commercial lures and positioned along the 12 km coast, less than 100 m from the shoreline. Another three traps were set in the interior, approximately 500 m from the coast.

RESULTS

Eradication of Norway rats has been declared successful on Grass Island at South Georgia and on 30 of the 39 islands baited between 2001 and 2009 in the Falklands (Table 1).

Treatment failed on seven islands, some of which were subsequently re-baited.

Tussacbirds have re-established on five islands cleared between 2001 and 2003 in the Falklands. There are anecdotal reports of an increase in the white-chinned petrel (*Procellaria aequinoctialis*) and sooty shearwater (*Puffinus griseus*) populations. There is evidence to suggest that the number of songbird species and the number of birds increases after eradication (D. Brown data; S. Poncet data; R. Woods pers. comm.), although there is no record of any island being re-colonised by Cobb's wrens.

South Georgia pipits have re-established on Grass Island at South Georgia, with anecdotal reports of an increase in the white-chinned petrel population.

Patagonian grey foxes have been eradicated from two islands in the Falklands.

Table 1 An inventory of island restoration operations between 2000 and 2009 in the Falklands and South Georgia.

Map Ref. in Fig. 1	Island Name	Area (ha)	Year treated, Supervisor, Organisation	Method	Status, year of last check
Norway rat *(Rattus norvegicus)*					
20	Grass Island (South Georgia)	30	2000, A Cox & I McFadden/ GSGSSI	Pestoff 20R; 10 kg/ha; hand broadcast	Rat-free 2008
5	Top + Bottom Islands	12 + 8	2001, D Brown/FC	Pestoff 20R; 1.6 kg/ha, 0.6 kg/ha; bait stations	Rat-free 2009
6	Outer, Double + Harpoon Island	22 + 9 + 3	2001, D Brown/FC	Pestoff 20R; 5.5 kg/ha, 5 kg/ha, 4.2 kg/ha; hand broadcast	Rat-free 2009
7	Rookery, Cucumber + Rat Islands	25 + 3 + 1	2002, N. Huin/FC	Pestoff 20R; 3.6 kg/ha, 8.6 kg/ha, 5 kg/ha; hand broadcast	Rat and Rookery rat-free 2008, re-inv. 2010; Cucumber rat-free 2010
8	North East, Hutchy's + Ella's Islands	305 + 12 + <4	2003, D Brown/FC	Pestoff 20R; 4.2 kg/ha; hand broadcast	Rat-free 2008
9	Pete's Islet	<1	2003, D Brown/FC	Pestoff 20R; 4.2 kg/ha; hand broadcast	Rat-free 2011
10	Outer North West Is.	65	2004, N. Huin/FC	Pestoff 20R; 6 kg/ha; hand broadcast	Failed or re-invaded 2007
11	South West Horse Is.	3	2005, N. Huin/FC	Pestoff 20R; hand broadcast	Failed or re-invaded 2011
12	Halt Island	13	2006, D Christie/ landowner	Pestoff 20R; 9 kg/ha; hand broadcast	Rat-free 2009
13	Inner North West Is. + islet	36.5 + 1.5	2007, N. Huin/FC	Pestoff 20R; 7.5 kg/ha; hand broadcast	Failed or re-invaded 2009
14	Channel east + west, Stick in the Mud, Skull Bay, Green, Coffin + islet & Letterbox Is.	21 + 26 + 3 + 7 + 24 + 23 + <1 + 3	2007, S Poncet/BILC	Ditrac; 10.5 kg/ha; hand broadcast	Rat-free 2009
15	Governor Island	270	2008, S Poncet/BILC	Ditrac; 10 kg/ha; hand broadcast	Rat-free 2010
4	Tea Island	320	2009, S Poncet/BILC	Ditrac; 10.3 kg/ha; hand broadcast	Rat-free 2011
16	Amy Is. + the Knobs	3.6 + 1 + <1	2009, S Poncet/BILC	Ditrac; 20 & 16 kg/ha; hand broadcast	Rat-free 2011
17	Sniper Island	3.4	2009, S Poncet/BILC	Ditrac; 21 kg/ha; hand broadcast	Rat-free 2011
10	Outer North West Is. (2nd attempt)	65	2009, L Poncet/FC	Pestoff 20R ; 10 kg/ha; hand broadcast	Pending
13	Inner North West + islet (2nd attempt)	36.5 + 1.5	2009, L Poncet/FC	Ditrac; 10 kg/ha; hand broadcast	Pending
18	Pitt Island	16	2009, S Poncet/BILC	Ditrac; 10 kg/ha; hand broadcast	Rat-free 2011
19	Big + Little Samuel Islands + 3 islets	50 + 25 + 1 + 1 + 1	2009, B. Summers/FC	Ditrac; 8 kg/ha; hand broadcast	Rat-free 2011
Patagonian grey fox *(Lycalopex griseus)*					
1	Sedge Island	330	1966-81, W McBeth	Shooting; trapping	Eradicated
2	Weddell Island	21850	1997-98, J & S Ferguson	1080; bait stations & aerial broadcast; shooting & trapping	Failed
3	Beaver Island	3800	1997-98, S Poncet	1080; shooting; bait stations	Failed
4	Tea Island	320	2008, R Ellis	Trapping	Eradicated
Feral cat *(Felis catus)*					
3	Beaver Island	3800	ca. 1986, T Felton	Shooting; trapping	Eradicated

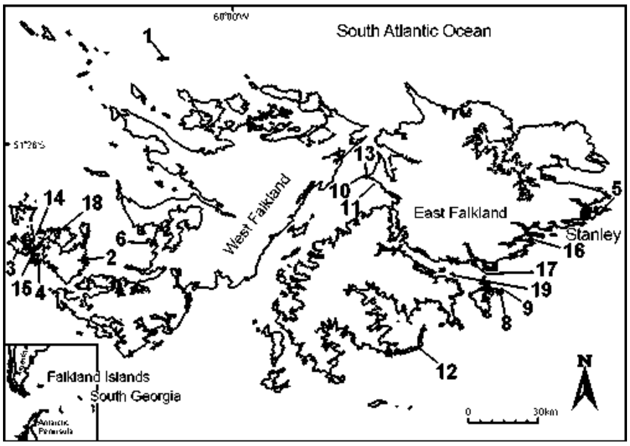

Fig. 1 The Falkland Islands, showing islands where rat eradication operations have taken place between 2001 and 2009. Names for the numbered islands are given in Table 1.

A core group of local operators with rat and fox eradication experience and skills is now in place.

Eradication projects since 1997 have created public interest in and awareness of the effects of introduced species and the benefits of eradication and biosecurity.

Case studies

1. Norway rats on Grass Island, South Georgia. This project was commissioned by the South Georgia Government in 2000, as part of a rat eradication feasibility study for South Georgia (Poncet et al. 2002). In 2000, Pestoff 20R was hand broadcast at 10 kg/ha over a 50 m grid. South Georgia pipits had returned to the site by 2003, with anecdotal reports of an increase in the white-chinned petrel population.

2. Norway rats on 21 islands in the Falkland Islands 2001-2009, using Pestoff 20R and Ditrac. These campaigns provided FC staff and volunteers with training. Restoff 20R was placed in bait stations on two islands (12 ha and 8 ha) and hand broadcast on another three (9 ha, 3 ha and 20 ha) (Brown *et al.* 2001). The operations were successful and within three years tussacbirds had returned to two of the five islands (Ingham *et al.* 2005; Forster 2007). A further 11 islands were baited between 2002 and 2007, including the largest island attempted at this time in the Falklands, North East Island (302 ha, with a baiting regime of 4.2 kg/ha). Of these 11 operations, five were successful (notably North East Island) and six either failed or the islands were re-invaded (Woods *et al*. 2003; Ingham *et al.* 2005; Poncet 2006; Forster 2007; S. Poncet data). Five islands were successfully treated with Ditrac in 2009.

3. Norway rats on 15 islands in the Falklands 2007-2009, using Ditrac wax blocks. These campaigns were designed by BILC and provided training and employment for the local community. Bait was hand broadcast on islands ranging in size from <1 ha to 320 ha. The nine islands baited in 2007 and 2008 were confirmed rat-free in 2009 and 2010 (S. Poncet data). The remaining six islands (which include one of 320 ha) were baited in 2009, and were rat-free in 2011.

4. Patagonian grey fox in the Falklands. Foxes were introduced from Argentina to Weddell Island in 1929 for fur-farming. Animals were further released on Beaver Island (3,800 ha), Tea Island (320 ha), Staats Island (500 ha), Split Island (220 ha), Sedge Island (330 ha) and River Island (450 ha) in the 1930s. The Sedge Island population was eradicated over a period of 15 years by the land owner/farmer using a combination of trapping, shooting and snares. An unsuccessful campaign to eradicate foxes on Beaver Island and Weddell Island in 1997 and 1998 used mainly 1080 poison (Foxoff, 3 ppm sodium fluoroacetate) supplemented by shooting and cage traps (Ferguson and Ferguson 1998; Poncet 1998). Traps and snares were used successfully in 2008, to eradicate foxes from Tea Island (320 ha). A total of 33 foxes were trapped, and after thorough checks in August 2009, the island was declared clear of foxes.

DISCUSSION

This past decade's efforts to eradicate Norway rats from offshore islands in the Falklands Islands are the fruition of the 2001 island restoration plan. The majority of islands treated were identified in Brown (2001), who also recommended the use of standardised biological surveys of islands, regular surveys to check for rodent presence following eradication and the establishment of a local group responsible for island management and restoration.

In 2008, the list of islands suggested as priorities in 2001 was reviewed at a rat eradication workshop organised by the South Atlantic Invasive Species Programme (Miller 2008). The revised list has been incorporated into the framework for prioritising future hand broadcasting operations. The procedures process was further refined in 2009, with the introduction of a rat eradication register (Excel format) for recording details of each operation, peer-reviewed pre-baiting surveys, and operational plans designed for each island's specific requirements and the type of bait available (Pestoff20R, Brodifacoum-25W Conservation, or Ditrac).

The success of rat eradications over the past decade in the Falklands has not only resulted in major ecological gains with the return of tussacbirds to 5 of the 41 islands treated and increases in small songbirds; it has had a positive impact on community understanding of island restoration and biosecurity. This has been further strengthened by the establishment of a core group of operators with the capacity to develop eradication techniques for local conditions and to participate in eradications at South Georgia. Furthermore, the use of local operators ensures that overseas funding for each project is spent within the Falklands. Expenditure on local employment, training opportunities, goods and logistics also increases community involvement and support for future eradications and biosecurity.

The apparent inability of Cobb's wrens to recolonise islands raises the question of whether flight distances from source populations are too great for the birds. In this event, translocation may be the only way to speed up the process.

Of highest concern however, is the risk of rats re-invading treated islands by swimming. The re-invasion of Rat Island and Rookery Island six years after successful treatment in 2002, may be evidence to suggest that rats first re-colonised Rat Island, 300 m from Beaver and then swam the 500 m to Rookery Island. The previously accepted 350 m maximum swim distance of rats in Falklands waters has been revised in the light of these incursions. Additionally, the rat status and separation distance of 208 islands in the Falklands were analysed in order to obtain more information on rat dispersal. Islands closer than 500 m to the nearest rat-infested land were found to have a 1 in 3 chance of being re-invaded; this decreased to 1 in 10 for islands further than 1 km, while the 50 islands that were over 2 km distant were rat-free (Martinez del Rio and Tabak pers. comm.). This information is now being used when assessing the suitability of islands for eradication. However, the various factors that cause rat incursions remain unknown.

Since 2009, a further 13 islands have been baited with either Pestoff 20R or Brodifacoum-25W Conservation pellets, bringing the total of islands treated to 52. One of the islands, First Passage (750 ha) is the largest island to have been treated by hand broadcasting. The South Georgia operation began in 2010 when Saddle Island and over 12,000 ha of the main island were baited by helicopter.

The following lessons were learnt from our experiences in the Falklands and South Georgia:

1. Operational plans based on pre-baiting surveys are essential for avoiding mistakes.

2. Familiarity with the eradication site is crucial for good planning.

3. Employ trained locals: a team of paid, locally based and experienced operators who are familiar with the environment reduces operator error, increases efficiency and provides skills and capacity for future eradication projects.

4. Specialist advice at all stages of planning and for every new situation is invaluable: the attempted fox eradication on Beaver Island in 1997-98 reduced the population to a few individuals but ultimately failed due to lack of funding, labour, specialist advice and momentum.

5. Ensure that checks for rodent presence are made once a year for at least two years following an eradication attempt in order to monitor for potential incursions.

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Eradication of exotic mammals from offshore islands in New South Wales, Australia

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Abstract Operations to eradicate populations of exotic mammals – ship rat (*Rattus rattus*), house mouse (*Mus musculus*) and European rabbit (*Oryctolagus cuniculus*) – have recently been conducted on five offshore islands in New South Wales, Australia. Techniques involved the broadcast and bait-station application of cereal baits containing the anticoagulant brodifacoum. Brush Island (47 ha) was treated for rats using bait stations in July 2005 and declared pest-free in 2007 after monitoring failed to detect any rodents. Recent observations have revealed increased numbers of lizards, frogs and crabs, as well as the presence of the white-faced storm-petrel (*Pelagodroma marina*), a species not previously recorded breeding there. Montague Island (82 ha) was aerially baited for mice and rabbits in July 2007 and declared free of these pests in 2009. The removal of these exotic mammals was undertaken primarily to enhance restoration of seabird nesting habitat following the removal of invasive kikuyu grass (*Pennisetum clandestinum*). Broughton Island (144 ha) was aerially baited for rats and rabbits in August 2009. The same operation included aerial baiting of nearby Little Broughton Island (30 ha) and Looking Glass Isle (4 ha) to remove rats. Detector dogs were used to search for surviving rabbits on Broughton Island in November 2009, but failed to detect any sign of them. One month later Gould's petrel (*Pterodroma leucoptera*) was recorded breeding there for the first time. Monitoring for the presence of rats and rabbits is continuing. Knowledge sharing and the free availability of information have been pivotal to the success of the operations undertaken to date, and the experiences gained have greatly enhanced our local capacity to plan and co-ordinate more complex eradications.

Keywords: Brodifacoum, eradication, house mouse, *Mus musculus*, island, kikuyu, *Pennisetum clandestinum*, rabbit, *Oryctolagus cuniculus,* ship rat, *Rattus rattus*

INTRODUCTION

Introduced mammals have had severe impacts on island systems, causing the extinction or local extirpation of numerous species worldwide (Groombridge 1992). Their eradication from islands has generally been highly beneficial for many ecosystem components including seabirds, terrestrial birds, lizards, amphibians, invertebrates and plant communities (Newman 1994; Towns and Broome 2003; Howald *et al*. 2007). The range of exotic mammals that established populations on offshore islands in New South Wales (NSW), Australia, includes goat (*Capra hircus*), pig (*Sus scrofa*), cat (*Felis catus*), European rabbit (*Oryctolagus cuniculus*), ship rat (*Rattus rattus*) and house mouse (*Mus musculus*). The larger of these species were eradicated from NSW islands between 1980 and 2000, after which only rodents and rabbits remained (Table 1).

Exotic rodents can have devastating impacts on island ecosystems (Towns *et al.* 2006; Jones *et al.* 2008) and have long been acknowledged as a significant threat to the native ecosystems of South Pacific islands (Atkinson and Atkinson 2000). Rats prey on the eggs and chicks of land birds and seabirds, and can cause major declines in these species (Merton *et al.* 2002). Rats and mice also prey heavily on reptiles, snails, insects and other invertebrates (Towns 1991; Bergstrom and Chown 1999; Smith *et al.* 2002; Hadfield and Saufler 2009) and compete with native avifauna for food (Huyser *et al.* 2000). They consume quantities of flowers, fruits and seeds, which can reduce seedling recruitment (Shaw *et al.* 2005), leading to loss of species and changes in vegetation communities (Auld *et al.* 2010). By reducing seabird abundance, rodents can reduce the inflow of marine-derived nutrients which, in turn, can profoundly affect the productivity of insular vegetation communities (Bancroft *et al.* 2005). On Lord Howe Island, rats are implicated in the extinction of at least five species of endemic birds, 13 species of invertebrates and two plant species (LHIB 2009), and are a continuing threat to at least 13 other bird species, two reptile species, 51 plant species, 12 vegetation communities and numerous species of threatened invertebrates (DECC 2007).

The impact of rabbits on islands worldwide has been catastrophic, with many islands being virtually denuded (Watson 1961; Clapp and Wirtz 1975; Coyne 2010). Impacts have been less severe in NSW, although loss of vegetative cover through rabbit grazing and burrowing activities has rendered substantial areas of some islands vulnerable to erosion and weed invasion.

In 1997, rabbits were successfully eradicated from Cabbage Tree Island on the central coast of NSW (Priddel *et al.* 2000) to protect and restore the habitat of the endangered Gould's petrel (*Pterodroma leucoptera leucoptera*), an endemic subspecies that breeds principally on this island (Priddel and Carlile 1997a). Rabbits had removed the rainforest understorey, allowing pied currawongs (*Strepera graculina*) easier access to the forest floor, where they hunted and killed nesting petrels and their chicks (Priddel and Carlile 1995). The removal of the understorey also allowed the sticky fruits of the birdlime tree (*Pisonia umbellifera*) to fall directly to the forest floor, increasing the likelihood of petrels becoming entangled in them (Priddel and Carlile 1997b). Entangled birds are often unable to fully open their wings to fly, and die from starvation. Rabbits were also restricting the regeneration of many rainforest canopy species (Werren and Clough 1991). For example, seedlings of the cabbage tree palm (*Livistona australis*) survived only if they were caged to prevent grazing by rabbits (Carlile 2002). Lack of seedling recruitment over the 90 years that rabbits were present threatened the continued survival of this species on the island.

Following the removal of rabbits from Cabbage Tree Island, vegetation regeneration was so extensive that, in 2003, the NSW Government initiated a programme to remove mammalian pests from all NSW offshore islands. At that time, the only islands in NSW known to have populations of exotic mammals were Brush Island, Montague Island, three islands within the Broughton Island group, South Solitary Island and Lord Howe Island (Table 1). Operations

Pages 337-344 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

Table 1 Populations of introduced mammals on NSW islands, and their eradication.

Island	Area (ha)	Spp targeted	Erad.	Method(s)	Year	Source
South Solitary	10	European rabbit	Yes	Shooting; myxomatosis	<1975	Lane 1975
Lord Howe	1455	Feral house cat	Yes	Shooting; trapping	1980	Miller and Mulette 1985
Lord Howe	1455	Pig	Yes	Shooting	1981	Miller and Mulette 1985
Bowen	50	European rabbit	Yes	Hand broadcasting of 1080-laced carrots; myxomatosis	1981	Martin and Sobey 1983
Montague	82	Goat	Yes	Shooting	1988	R.Constable (pers.comm.)
Bowen	50	Ship rat	Yes	Bait stations (50x50 m grid) containing bromadiolone (50 ppm) or brodifacoum (50 ppm) in wax blocks	1993–1995	Meek 2009
Cabbage Tree	26	European rabbit	Yes	Myxomatosis, rabbit haemorrhagic disease, aerial dispersal of brodifacoum (50 ppm) in cereal pellets	1997	Priddel *et al*. 2000
Lord Howe	1455	Goat	Yes[1]	Shooting	1999	Parkes *et al*. 2002; Priddel and Hutton 2010
Brush	47	Ship rat	Yes	Bait stations (25x25 m grid) with brodifacoum (50 ppm) in wax blocks	2005	This study
Montague	82	House mouse; rabbit	Yes	Natural outbreak of rabbit haemorrhagic disease; aerial dispersal of brodifacoum (20 ppm) in cereal pellets; hand-baiting buildings	2007	This study
Broughton	144	Ship rat; rabbit	Yes	Rabbit haemorrhagic disease; aerial dispersal of brodifacoum (20 ppm) in cereal pellets; hand-baiting in and around buildings	2009	This study
Little Broughton	30	Ship rat	Yes	Aerial dispersal of brodifacoum (20 ppm) in cereal pellets	2009	This study
Looking Glass	4	Ship rat	Yes	Aerial dispersal of brodifacoum (20 ppm) in cereal pellets	2009	This study
South Solitary	10	House mouse	No	Aerial dispersal of brodifacoum (20 ppm) in cereal pellets	Planned	
Lord Howe	1455	Ship rat; house mouse	No	Aerial dispersal of brodifacoum (20 ppm) in cereal pellets; hand-baiting and bait stations in settlement area	Planned	

[1] a few females remained after 1999 but these have since died out

to eradicate these pests have recently been conducted on all these islands except South Solitary and Lord Howe. In this paper, we document the procedures used, along with any observed non-target impacts, outcomes and biodiversity benefits. We also highlight some challenges encountered and discuss information gaps.

STUDY SITES

Brush Island

Brush Island (35°31´S, 150°25´E; 47 ha) is a nature reserve situated 370 m offshore from Bawley Point, 23 km south of Ulladulla on the NSW south coast (Fig. 1). Ship rats were common throughout the island and probably arrived when a steamer, the *Northern Firth*, ran aground in 1932. Direct human disturbance on Brush Island is limited, with little visitation and no record of recent human habitation.

Montague Island

Montague Island (36°15´S, 150°13´E; 82 ha) is a nature reserve situated 7 km offshore, 10 km southeast of Narooma on the NSW south coast (Fig. 1). The island is volcanic in origin, and comprises two sections (a southern section and smaller northern section), divided by a deep ravine. A building precinct, located at the highest point on the southern section, contains a lighthouse and accommodation built in 1881 for three lighthouse keepers and their families, as well as a number of outbuildings and associated infrastructure. Nowadays, the lighthouse is automated and the buildings are used as a museum and accommodation for maintenance workers, visiting scientists and eco-tourists.

The island once supported small trees, but the combined effects of timber harvesting for construction and fuel, the increased frequency of wildfire, and grazing by rabbits and goats have resulted in the loss of most woody vegetation (Heyligers and Adams 2004). Presently, the dominant

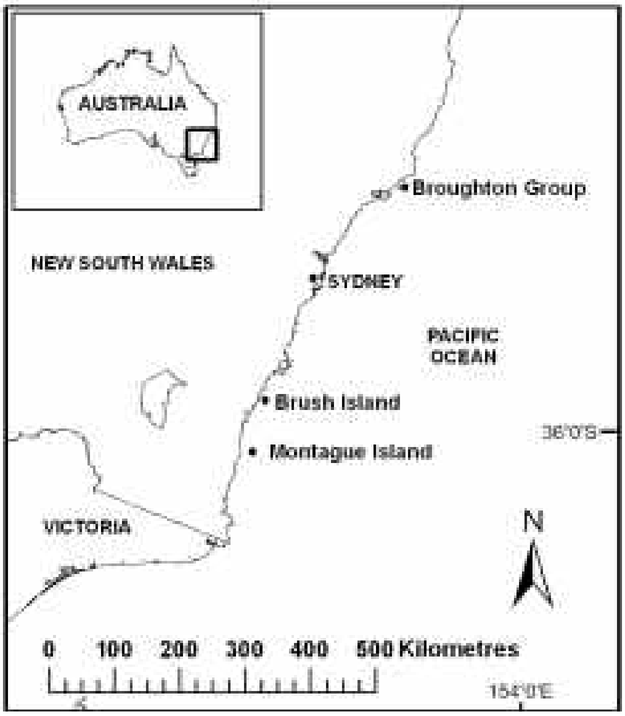

Fig. 1 Location of NSW islands where eradication operations were undertaken.

vegetation is spiny-headed mat-rush (*Lomandra longifolia*), bracken (*Pteridium esculentum*), coastal tussock grass (*Poa poiformis*) and introduced kikuyu grass (*Pennisetum clandestinum*). Kikuyu was spreading rapidly, and by 2001 it covered more than a third of the island (Weerheim *et al.* 2003). Since that time, an ongoing control programme has removed the majority of this invasive weed. Areas from which kikuyu has been removed have been replanted with native seedlings grown at local nurseries from local seed stock.

Mice were present throughout the island at densities varying from 59–160 per ha (Cory 2007), and all buildings were heavily infested. Intermittent control using rodenticide and snap traps had been attempted in and around buildings, with limited success. Rabbits were common, particularly around the rocky fringes of the island. Although the myxoma virus occurred on the island, as evidenced by periodic outbreaks of myxomatosis, it was seldom very effective, probably because the rabbits were largely surface dwelling and rabbit fleas (a prime vector) were not present (Silvers and Davey 1994). The only attempt at controlling rabbits was the periodic use of 1080-laced carrots before 1995.

In 2005, the numbers of rabbits dropped dramatically due, we believe, to an outbreak of rabbit haemorrhagic disease caused by the natural spread of calicivirus. At the time of the eradication operation in 2007, rabbits were in low numbers and, as far as could be ascertained, were confined to the northern section of the island.

Broughton Island group

The Broughton Island group is situated approximately 3 km offshore and 15 km northeast of the entrance to Port Stephens on the NSW central coast (Fig. 1). The group is volcanic in origin and consists of five islands totalling 182 ha: Broughton Island (144 ha), Little Broughton Island (30 ha), Looking Glass Isle (4 ha), North Rock (3 ha) and Inner Rock (1 ha). Broughton Island, the main island in the group, is part of Myall Lakes National Park; the other islands are nature reserves. Rats were present on Broughton Island, Little Broughton Island and Looking Glass Isle. As far as is known, rabbits occurred only on Broughton Island.

Broughton Island (32°36´S, 150°19´E), has been used as a base for commercial fishing since the mid 19th century (Clarke 2009). Two small settlements were established soon after the First World War; one of which was abandoned in 1939; the other, now a hamlet of seven huts, is occupied by recreational fishers and their families on a semi-permanent basis, with up to 50 persons present at any one time.

Rainforest once existed on the higher slopes of the island, but occupation led to a marked increase in the frequency of fires (Lane 1976) as fishermen would burn the island to control undergrowth and clear tracks (Clarke 2009). The increase in fire frequency has reduced the amount of woody vegetation, such that only scattered trees now remain (Lane 1976). The island supports a large and important population of green and golden bell frog (*Litoria aurea*), a species confined to southeastern Australia and listed as threatened in NSW (White and Pyke 1996).

Rabbits were taken to Broughton Island in 1906 when the Danysz Rabbit Inoculation Station was established on the island to investigate the potential for a new strain of *Pasteurella* bacterium to control rabbit numbers on the Australian mainland (Hindwood and D'Ombrain 1960). Unfortunately, although capable of killing rabbits, the bacterium failed to propagate through wild populations and, after twelve months, the project was abandoned (Rolls 1969). Subsequently, rabbits were trapped and shot for food by the island's inhabitants, but as far as we can ascertain the only attempted control was the introduction of myxoma virus some time after 1950.

Nothing is known about when or how rats came to Broughton Island; they were known to be present in the 1960s but probably arrived much earlier. In recent decades, rats were common within the vicinity of the huts, where they regularly contaminated foodstuffs. Their impact on nesting seabirds has never been investigated, but they are presumed responsible for the local extirpation of the white-faced storm-petrel (*Pelagodroma marina*), a species that is numerous on the outer islets of North Rock and Inner Rock. Control of rodents has been limited to activities in and around buildings, using rodenticide and snap traps.

Little Broughton Island (32°37´S, 150°20´E) is separated from the main island by a deep narrow channel. Much of the island is dominated by mat-rush although the peak is heavily wooded with coastal tea-tree (*Leptospermum laevigatum*), tuckeroo (*Cupaniopsis anacardioides*) and coast banksia (*Banksia integrifolia*). Access to the island is difficult and it is seldom visited. A brief inspection of the island in 1998 found rats to be particularly abundant, as evidenced from an exceptionally high density of droppings and a marked browse line about 15 cm above ground, below which all edible vegetation had been removed.

Looking Glass Isle (32°37´S, 150°19´E) is rocky and steep-sided. The dominant vegetation is ruby saltbush (*Enchylaena tomentosa*), mat-rush and the introduced prickly pear (*Opuntia* sp.). The presence of droppings in 2009 indicated rats were present. At low tide it is possible to wade between this isle and Broughton Island.

North Rock (32°35´S, 150°19´E) and Inner Rock (32°35´S, 150°18´E) are both vegetated, but there is no record of exotic mammals on either of these islets. However they are only 1.4 km and 0.5 km, respectively, from Broughton Island, well within the swimming range of rats. Public access to these islets is prohibited.

Potential non-targets

The only native mammals present on NSW offshore islands are fur seals (*Arctocephalus* spp.), frugivorous megabats (Megachiroptera) and insectivorous microbats (Microchiroptera). These animals are highly unlikely to consume cereal baits and thus were not considered to be at direct risk of rodenticide exposure. Seabirds (petrels, shearwaters and terns) occur on those islands where eradication operations were conducted, but were not considered to be at risk due to their piscivorous diet. Silver gulls (*Chroicocephalus novaehollandiae*) breed on some islands, but were absent during the time that baits were present.

The only land bird likely to consume baits was the buff-banded rail (*Gallirallus philippensis*), a nomadic species that fluctuates in abundance. At times, there have been up to 20 rails recorded on Montague Island, but when baiting was conducted, only two individuals were observed. Several raptors were potentially vulnerable to secondary poisoning by consuming contaminated rabbits and rodents, but all occurred in low numbers: white-bellied sea-eagle (*Haliaeetus leucogaster*), swamp harrier (*Circus approximans*), peregrine falcon (*Falco peregrinus*) and Australian kestrel (*F. cenchroides*).

Brodifacoum, an anticoagulant, was not expected to have significant effects on invertebrates as these organisms have different blood clotting systems to mammals and birds. Although invertebrates may feed on the bait, insectivorous birds and bats were not considered to be at risk because invertebrates are unlikely to accumulate high levels of brodifacoum as it is quickly eliminated through metabolism and excretion (Morgan *et al.* 1996). Very large numbers of contaminated invertebrates would need to be consumed in a relatively short period to cause mortality of insectivorous bats and birds (Morgan and Wright 1996).

METHODS

Brush Island

The eradication of rats from Brush Island was conducted in 2005 using bait stations constructed from 35 cm lengths of flexible drainage pipe (10 cm diameter). A 25x25 m grid was established across the entire island and a single bait station was placed at each of the 550 grid points. Rodenticide bait (Pestoff Rodent Blocks, Animal Control Products, Wanganui, New Zealand) containing the anticoagulant brodifacoum at 50 parts per million (ppm) was added to each bait station on 7 July 2005 (Day 0). Three of these wax blocks, each about 30 g, were threaded onto a short length of wire tied into each bait station. Baits were replenished approximately every second day for the first 10 days, with approximately half of the stations serviced on any one day. Bait stations were then inspected approximately 2, 4 and 8 weeks later and replenished (to approximately 90 g) as required. At each inspection, the weight of bait remaining in each bait station was recorded along with the weight of bait added. The total amounts of bait used and consumed during the operation were calculated. Carcasses found during baiting operations were removed to reduce the risk of secondary poisoning of non-targets. In October 2005, the bait stations and remaining bait were removed.

Monitoring to detect for the presence of rats was undertaken over a period of six weeks in late 2007. A total of 50 feed stations containing a measured number of non-toxic cereal pellets (Pestoff Rodent Bait 20R, Animal Control Products, Wanganui, New Zealand) were randomly distributed across the island. These stations were checked approximately weekly, and any loss of pellets recorded.

Montague Island

The eradication operation on Montague Island was conducted during winter (July 2007) when mouse densities were seasonally low and after rabbit numbers had been reduced substantially, probably by a natural outbreak of haemorrhagic disease. The operation involved two aerial applications of cereal-based bait (Pestoff Rodent Bait 20R) containing brodifacoum at 20 ppm. To investigate the efficacy of bait size in eradicating mice, the southern section of Montague Island was baited with 10 mm baits (~2 g pellets), and the northern section with 5.5 mm baits (~0.6 g pellets). There is sufficient brodifacoum in one small pellet to kill a mouse. Sowing rates for both sizes were 12 kg per ha for the first drop and 6 kg per ha for the second. The second application took place 10 days after the first.

Bait was delivered using a spreader bucket slung below a helicopter (Eurocopter AS350B3) equipped with a GPS navigation and guidance system (AG-NAV® Guía). The bucket provided an effective swathe width of 80 m for 10 mm bait and 70 m for 5.5 mm bait. Parallel flight lines were spaced at 35 m intervals for 10 mm bait and 30 m for 5.5 mm bait, giving a swathe overlap in excess of 50%. A 30 m exclusion zone around the building precinct was baited by hand. Bait stations were placed in each room of each building and in all accessible roof cavities. There were no under-floor spaces.

One month after the second baiting, 75 tracking tunnels (Connovation, Auckland) were strategically distributed alongside tracks on the island. Tunnels were monitored for mouse activity (footprints) and sampled at approximately 3-month intervals for 24 months. At each visit, new ink boards and attractant (linseed oil) were fitted to each tunnel, since their effective life was limited to about two weeks. In addition, up to 100 Elliott traps (baited with peanut butter and oats) along with seven remotely activated cameras were deployed near any reported sightings of mice. We also looked for fresh rabbit dung, grazing and diggings while conducting other work on the island. As a biosecurity measure, seven permanent bait stations have been set up on the island; these are also monitored for activity (and replenished if necessary) every three months.

Broughton Island group

Beginning in 2009, the rabbit population on Broughton Island was reduced using rabbit haemorrhagic disease. The virus, sprayed onto diced carrots, was distributed around the island on 15 April 2009; almost four months before baiting took place. This was done to minimise the likelihood of secondary poisoning of raptors (by reducing the number of poisoned dead or dying rabbits), and to increase the amount of bait available for consumption by rats, as well as the remaining rabbits.

In August 2009, all islands within the Broughton group were aerially baited twice with 10 mm Pestoff Rodent Bait 20R. Although rats and rabbits were not known to occur on North Rock or Inner Rock, as a precaution these vegetated islets were also baited. Each application was sown at the rate of 12 kg per ha. The second application took place 14 days after the first. Bait was delivered aerially using the same equipment and techniques used on Montague Island, except that swathe overlap was reduced to 50% (i.e. 80 m swathe and 40 m flight lines). A 30 m exclusion zone around the building precinct was baited by hand. Bait stations containing five pellets of Pestoff 10 mm bait were placed under and within all buildings, including in all accessible roof and under-floor cavities. These were replenished after 14 days and removed after 100 days.

Although brodifacoum is insoluble in water and is not known to affect frogs, as a precaution all pools known to contain green and golden bell frogs were monitored continuously throughout each bait drop, and any baits that fell into these pools were removed immediately.

During two days in November 2009, three trained detector (sniffer) dogs were used to search for surviving rabbits. An island-wide survey to search for fresh rabbit dung, grazing and diggings was also undertaken at this time. In November 2009, 30 tracking tunnels were randomly distributed across the island and six wax tags (Pest Control Research, Christchurch) were deployed around the buildings. These devices will be left in place and inspected quarterly until August 2011, when another island-wide survey will be conducted. Eight permanent bait stations have been set up around the huts and these will be monitored for activity (and replenished if necessary) every 3 months. If there is no evidence to the contrary, the operation will be declared a success in August 2011.

RESULTS

Brush Island

A total of 123 kg of bait was placed into the bait stations, of which 84 kg was consumed by rats. Rats began taking bait immediately, and more than 98% of total bait consumption occurred within the first 7 days. The remaining 2% was taken between Day 8 and Day 10. Although baits were checked periodically over the following three months, there was no further evidence of bait take and no sign of rats being present. The first dead rats appeared four days after baiting commenced and no fresh carcasses were found after Day 10. Judging from the bait take and carcasses found, the majority of rats died within the first week.

The eradication on Brush Island was declared a success in 2007 after monitoring failed to detect any rats. No pellets showed any sign of being gnawed and none were removed from any of the 50 feed stations during the monitoring period.

Anecdotal observations during subsequent visits have revealed an apparent increase in the numbers of southern water skink (*Eulamprus heatwolei*) and two species of amphibians: striped marsh frog (*Limnodynastes peronii*) and the eastern common froglet (*Crinia signifera*). Purple rock crabs (*Leptograpsus variegatus*) are also noticeably more common and the average body size appears to have increased. Two years after the eradication operation the white-faced storm-petrel was recorded burrowing on the island for the first time. These diminutive birds (~60 g) are highly vulnerable to rats and are likely to have bred on the island before rats arrived. The island's flora also appears to be recovering with, for example, banksia seedlings now much more prevalent.

Montague Island

Monitoring of the tracking tunnels during the 24 months after baiting failed to detect any mice, and none have been seen in any of the buildings. During the same period, surveys have failed to find any evidence of rabbits on the island. Several reports were received of a small black mammal being seen on the island, but tracking tunnels, cameras and traps failed to find any corroborative evidence. We now believe that these sightings were of buff-banded rail chicks. Montague Island was declared free of mice and rabbits in July 2009. The successful eradication of mice from both sections of Montague Island demonstrated that bait size was not crucial in this instance.

Broughton Island group

Three months after the baiting operation, trained detector dogs did not find any sign of surviving rats or rabbits. The concurrent island-wide survey also found no evidence of either species. To date (December 2010), the tracking tunnels, wax tags and permanent bait stations have not detected any evidence of rabbits or rats, and none have been sighted around the buildings.

In December 2009, a single Gould's petrel was found incubating an egg on Broughton Island. This is the first record of this species breeding on this island. Previous searches of the one small area of suitable breeding habitat (rock scree) had found birds ashore, but there had been no evidence of breeding. Presumably, rats had destroyed any eggs, and the removal of this predator may facilitate the establishment of a population of Gould's petrel on the island.

Non-target impacts

Apart from an independent study of the green and golden bell frog, no monitoring of potential non-target species was undertaken, so results are mostly limited to anecdotal observations. An osprey (*Pandion haliaetus*) – a rare and threatened species in NSW – was killed in a collision with the helicopter distributing bait on Broughton Island. However, there was no significant difference in the number of raptors (individuals and species) present on Broughton after baiting compared to immediately before. Similarly, there has been no change in the number of green and golden bell frogs. On Montague Island, the only other island where some monitoring of avifauna has been conducted (Fullagar *et al.* 2009 and references therein), there has been no noticeable decline in the numbers or variety of raptors, despite the removal of all mammalian prey. Buff-banded rails were present on Montague Island in March 2007, not seen the following year, but were again present in 2009. It is possible that the baiting may have killed the few birds present and the species subsequently re-established. However, no dead birds were found, and annual surveys conducted during the seven years prior to baiting had failed to detect buff-banded rails on three occasions (43%).

DISCUSSION

At the time of writing (December 2010), all five eradications appear to have been successful, with no sign of exotic mammals on any of the islands treated. However, as detection of any small relic population is exceedingly difficult, the Broughton operation cannot be declared a success until August 2011, when final checks will be completed. By this time, two years after baiting, the target species would have increased in distribution and abundance such that it would be readily detectable. Meanwhile, the fact that trained dogs did not detect rabbits and the absence of teeth marks on wax tags are encouraging signs that this group of islands may now be free of exotic mammals.

Rabbits have been successfully eradicated from at least two, and most likely three, NSW islands – Cabbage Tree, Montague and Broughton – using brodifacoum baits as the primary mortality agent after populations had been reduced through disease. This combination of techniques has been an efficacious and cost-effective method of removing rabbits from NSW islands. In operations conducted elsewhere, however, poisoning has not been effective in eradicating rabbits, with some having survived the baiting operation for reasons that are not fully understood (Merton 1987; Jansen 1993; Torr 2002).

Conservation benefits

The biodiversity outcomes of removing exotic mammals from NSW islands have not been quantified; to date most information is largely anecdotal. Recent observations on Brush Island have revealed apparently increased numbers of lizards, crabs and frogs, as well as the presence of a seabird not previously recorded there, suggesting that rats were suppressing the numbers of these species. Vegetation on Brush Island also appears to be responding to the removal of rats, with unusually prolific seed and fruit production on many plants as well as a flush of young seedlings. Vegetation communities may eventually benefit from increased quantities of nutrients brought ashore by increased numbers of breeding seabirds following their release from rat predation (Fukami *et al.* 2006).

In 2001, large tracts (37%; Weerheim *et al.* 2003) of Montague Island were covered by a dense mat (~1 m thick) of introduced kikuyu grass, but a long-term programme to eradicate this invasive species has reduced its extent considerably. Areas from which kikuyu has been removed have been replanted with native seedlings. This initiative has seen large areas of the island transformed from a monoculture of kikuyu to more biodiverse native vegetation communities. While present, mice and rabbits were slowing the re-establishment of native vegetation by grazing seedlings and consuming seeds. With these pests gone, the process of natural regeneration is expected to accelerate.

The white-faced storm-petrel breeds on several rodent-free islands along the NSW coast; thousands once bred on Broughton Island (Hull 1911) but disappeared after rats arrived (Hindwood and D'Ombrain 1960). Storm-petrels are among the smallest petrels and are particularly prone to predation by rodents (Towns *et al.* 2006). Now that rats have been removed, these birds have already colonised (or recolonised) Brush Island and it is likely that they will also return to breed on Broughton and Montague islands.

The lack of mice infesting houses, contaminating foodstuffs and destroying equipment on Montague Island has provided significant social benefits as well as enhancing the protection and preservation of historically significant buildings. Similarly, the removal of rats on Broughton Island has ended a long battle by fishers to exclude rats from buildings and food stores.

Operational challenges

Planning the eradications described herein relied heavily on published information and the collective experience of practitioners worldwide, as well as advice from suppliers of equipment and materials. Knowledge sharing and the availability of information have been pivotal to the success of operations undertaken in NSW. The most appropriate poison to use, the type of bait, and the techniques of distributing bait were all well documented and readily transferrable. Certain other aspects of the operation, however, were less prescriptive and required adaptation to suit the specific biology of each island. These included the optimal sowing rate (particularly for operations targeting more than one species) and the efficacy of bait of different sizes (especially for eradicating mice). Other aspects that we also needed to address were: i) the possible destruction of baits by heavy rain soon after baiting; ii) the consequent need to undertake an additional bait drop to replace these rain-damaged baits; iii) the requirement to have in reserve additional bait to undertake such a contingency drop; and iv) the disposal of surplus bait.

Sowing rate in aerial operations is one of the most crucial aspects of the eradication programme. If too little bait is used then all individuals of the target species may not encounter the rodenticide or consume a lethal quantity, thus causing the eradication to fail. Too much bait increases costs and unnecessarily puts additional poison into the environment. Where practicable, trials with non-toxic bait, impregnated with a bio-marker, during the planning phase of the operation can provide useful information about the quantity of bait required. For a baiting operation to be effective, bait should be available to the target animal for at least 3–4 days. The rate at which baits are removed is dependent on the type and density of potential consumers present.

For the eradication on Montague Island, we opted to use sowing rates of 12 and 6 kg per ha for the first and second drop respectively. These were higher than have been used successfully elsewhere (Broome 2009) but were deliberately set high because of the presence of rabbits, a relatively large mammal capable of consuming large quantities of bait, thereby denying mice access to it. Reduction in the density of rabbits, possibly through disease in the months before baiting, reduced the potential competition for bait.

For the Broughton group, we again opted to use a sowing rate of 12 kg per ha for the first drop and 6 kg per ha for the second. This time, however, we also purchased an additional 6 kg per ha as a reserve to re-sow any areas not covered adequately due to equipment malfunction or error in application. Rather than remove and dispose of any unused portion of this reserve, we elected to distribute it on the island as part of the second drop. The total sowing rate was therefore 24 kg/ha (12 kg per ha for each of the two drops). To have the flexibility to drop additional bait in this way it is important, through careful planning and forethought, to ensure that all permits and approvals include such provision.

To avoid the issue of heavy rain soon after baiting and the consequent need to undertake any additional bait drop, baiting was conducted only when a week of fine weather was predicted. This restriction was not a problem for NSW where the weather is generally fair and reasonably predictable, but may be more difficult elsewhere.

Operations to eradicate mice have experienced higher rates of failure than rat eradications, potentially linked to inadequate bait coverage and encounter rates (MacKay *et al.* 2007; Howald *et al.* 2007). However, this theory has not been adequately investigated. Mice typically have smaller home ranges than rats, and therefore have a lower probability of being exposed to bait. To overcome these challenges during a bait-station operation, smaller spacing between stations can be used. For aerial operations, bait coverage can be enhanced by either increasing the quantity of bait distributed (kg/ha) or by reducing the size of the bait pellet. For any specific sowing rate, the smaller the pellet the greater the number of individual pellets broadcast. On Montague Island both 10 mm baits (~2 g pellets) and 5.5 mm baits (~0.6 g pellets) were used, and both successfully eradicated mice.

There were some disadvantages associated with using 5.5 mm bait for the aerial operation. Whereas the 10 mm bait was easily visible from within the helicopter, the 5.5 mm bait was much more difficult to see when broadcast, especially on poorly contrasting substrates. Verification that bait was being broadcast required an observer in the helicopter. Another problem with the smaller bait was that it billowed from the top of the spreader bucket. We remedied this problem by fitting a transparent cover over the top of the bucket; this prevented billowing but still allowed the pilot and observer to see the quantity of bait remaining in the bucket.

Capacity building

To build local eradication capacity we opted not to engage an interstate or overseas helicopter company or pilot with previous experience in eradication operations involving aerial application of bait. Instead, aerial baiting was undertaken using helicopters and pilots from the NSW National Parks and Wildlife Service. Although these pilots and the assisting ground crew were highly skilled in all kinds of helicopter work, including pest control and agricultural spraying, they had no previous experience in eradication. Eradication operations are very different from control operations and thus require a different mindset (Cromarty *et al.* 2002), with all individuals of the targeted species needing to be exposed to bait. Instilling and maintaining this mindset in all participants throughout all aspects of the operation proved to be a considerable, but surmountable, challenge.

As far as is known, and assuming the Broughton operation is successful, South Solitary Island (10 ha) and Lord Howe Island (1455 ha) are now the only NSW islands with exotic mammals (house mice and ship rats) still present. Eradication planning is currently underway for both these islands. Ship rats do occur on Muttonbird Island, Coffs Harbour, but this is not a true island as it is now connected to the mainland by a man-made breakwater. Lord Howe Island is an oceanic island situated 580 km east of the Australian mainland and 1570 km northwest of New Zealand. It is a World Heritage Area containing a large number of endemic plants and animals threatened by the presence of exotic rodents (DECC 2007). Aside from a number of non-target issues, any eradication operation on this island is complicated by the presence of a human population of ~350 permanent residents in 150 households, as well as livestock, pets, and a well-developed tourist industry.

A draft plan for eradication of exotic rodents on Lord Howe Island has been prepared (LHIB 2009), peer reviewed and released for public comment (*http://www.environment.nsw.gov.au/resources/pestsweeds/draftLHIrodentplan.pdf*). This operation, the first on an island with a large permanent population, is complex and will require continuing input from a broad spectrum of experienced planners and practitioners if it is to be successful. However, the experiences gained from the eradications reported in this paper have greatly enhanced our local capacity to plan and co-ordinate such an operation.

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Howland, Baker and Jarvis Islands 25 years after cat eradication: the recovery of seabirds in a biogeographical context

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Abstract Feral cats (*Felis catus*) were eradicated from Howland, Baker and Jarvis Islands, all U.S. National Wildlife Refuges in the equatorial Central Pacific Ocean, by the 1980s. The cats were introduced during the 1930s to control rodents, and succeeded in extirpating Norway rats (*Rattus norvegicus*) and Pacific rats (*R. exulans*), but not mice (*Mus musculus*). The cats also extirpated small species of seabirds including grey-backed terns (*Onychoprion lunata*), blue noddies (*Procelsterna cerulea*), brown noddies (*Anous stolidus*), Christmas (*Puffinus nativitatis*) and Tropical (*Puffinus bailloni dichrous*) shearwaters and Polynesian storm-petrels (*Nesofregetta fuliginosa*), and ate chicks and adults of larger Pelecaniformes such as boobies and frigatebirds. With cats eradicated, the extirpated seabirds began to recolonise these islands. Grey-backed terns recolonised Jarvis Island within four years and 20 years later Polynesian storm-petrels were also thought to be breeding there. On Baker and Howland Islands, which are separated by less than 40 miles, military occupations, invasive plants, and pests, apparently made some species of birds move between the islands. However, by 1996, the seabird diversity and population levels were returning to historically recorded levels. The position of these three islands near the Equator and in the flow path of the Equatorial Undercurrent (EUC), a cold, nutrient-rich subsurface current, has enhanced the recovery of seabirds. Regional and local upwelling provides nutrients which fuels high productivity of zooplankton, the primary food of blue noddies. On occasions, severe climate-driven fluctuations can impede upwelling and deprive marine ecosystems of nutrients that in turn affects seabird productivity. The strongest El Niño on record occurred during the 1982 cat eradication effort on Jarvis Island, suppressing seabird populations and thereby helping to limit cat numbers.

Keywords: Feral cats, seabird restoration, Equatorial Undercurrent, El Niño, Central Pacific Ocean, eradication outcome, marine monuments

INTRODUCTION

This paper records the responses of seabirds to cat eradications at Howland, Baker, and Jarvis Islands, in the context of regional biogeography over a 25 year period.

These three small, isolated desert islands are located within 48 nautical miles (nm) of the equator (Fig. 1) and have experienced the destructive effects of introduced rodents, cats (*Felis catus*), and plants, guano mining, and military encampments. Following their protection (in 1973) as United States national wildlife refuges, personnel of the Pacific Reefs National Wildlife Refuge Complex began to restore these islands. In 2009 these islands were designated as part of the Pacific Remote Islands National Marine Monument, providing protection of the surrounding waters extending out to 50 nm (Bush 2009).

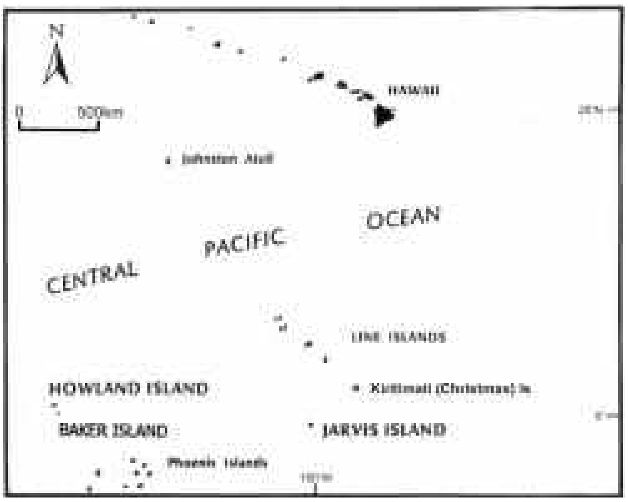

Fig. 1 Howland, Baker, and Jarvis Island locations in the Central Pacific Ocean.

Jarvis Island (0°23'S, 160°.01'W) (Fig.1) is the largest of the three islands with 445 ha of land. It is about 990 nm east of Howland and Baker Islands. Howland Island (0°49'N, 176°38'W) has an area of 162 ha and Baker Island (0°13'N, 176°31'W) has 138 ha of land. All islands are in a part of the tropical central Pacific Ocean where only low-lying atolls and submerged reefs occur.

The islands lie in the Equatorial Dry Zone. The nearest weather station is Kiritimati (Christmas) Island (Fig. 1) (USFWS 1998) where average monthly rainfall is approximately 75mm (range 0-500mm) per month, with precipitation consistent throughout the year (NOAA 1991). Howland, Baker and Jarvis are closer to the equator and drier than Kiritimati, in part because convective heating of these desert islands repels rain squalls.

All three islands share a common human history, as well as geography. Howland was visited by ancient Polynesian voyagers, as evidenced by an introduced population of kou trees (*Cordia subcordata*) and Pacific rats (*Rattus exulans*). It is likely that Baker and Jarvis were also visited (Hutchinson 1950), but the islands were too dry for permanent habitation. Whaling ships visited these islands in the early part of the 19th Century and ships were wrecked on them, probably introducing Norway rats (*R. norvegicus*) (Hague 1862).

Knowledge of historical numbers and species of wildlife is limited to a few historical accounts by early visitors, who noted the great abundance of seabirds and mined vast amounts of guano (Table 1). After more than a century of ecological degradation, small ground-nesting seabirds were extirpated and there was reduction in abundance of all birds (Hague 1862).

Cats were introduced to all three islands in 1936 for rodent control. They did not survive on Baker, and were eradicated from Howland by 1986 and Jarvis by 1990 (Table 1). Rats died out on all three islands, but house mice (*Mus musculus*) remain (Table 1).

Pages 345-349 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

Table 1 Historical timeline of introduction and eradication of predators, and selected human activities at Howland, Baker and Jarvis Islands.

Year	Howland	Baker	Jarvis
Pre-history	*Rattus exulans* introduced		
Early 1860s	Guano miners and whalers brought rodents. Species and islands not specified		
1858 - 1878	104,000 tons guano taken	300,000 tons guano taken	300,000 tons guano taken
1935	All three islands colonised, cats introduced		
		Norway rats named as present	
Post WW II	Cats probably exterminated Pacific rats	Cats probably exterminated Norway rats. Mice remain	
1963 -– 64	Cats removed from these two islands		211 cats killed (80% of popn)
1965	Cats allegedly introduced to these two islands by military		
1965		Mosquitoes introduced, island sprayed with DDT	
1982	Cats present	Cats died out naturally by now	118 cats killed (99% of popn)
1986	Final 17 cats killed		
1990			Last cat killed
2010	No introduced predators		Mice still present

METHODS AND MATERIALS

During irregular visits bird counts were non-standardised and were dependent on the number of observers and time on the island. Usually, there was insufficient time to do a complete census: only a description of which species were breeding and estimated numbers. At other times, teams walked abreast across swaths of land, tallying all species and numbers seen, until the island was completed surveyed. During the spring, when the largest numbers of birds were breeding, it took three to four days and nights for two people to count most of the birds, and in the autumn at one to two days to cover an island. Observer techniques, flock seasonality and El Niño events confounded estimation of sooty tern numbers.

RESULTS

Jarvis Island

On Jarvis Island the diversity of seabirds changed from 6-7 breeding species in 1982 to 14-15 species breeding in 2004. The species diversity has doubled and is now a full seabird community (Table 2).

The removal of most cats from Jarvis in 1982 (and the last one by 1990) was followed by a rapid increase in numbers of ground-nesting lesser frigatebirds (*Fregata ariel*). By 2004, there were two large lesser frigatebird colonies estimated to contain about 4000 birds. Colony phenology was variable; with birds at one colony beginning courtship whilst those at the other had post-fledging chicks.

Table 2 Seabird counts at the time of cat eradication for Jarvis, Baker, and Howland Islands and subsequent seabird counts on each island several years after cat eradication. The numbers represent the largest count of birds documented on a single trip but not the total population, as birds nest throughout the year.

Scientific Name	Common Name	Jarvis 1982	Jarvis 2004	Baker 1965	Baker 2002	Howland 1986	Howland 2007
Phaethon rubricauda	Red-tailed tropicbird	2500	2500	15	72	122	496
Sula dactylatra	Masked booby	3000	7000	400	3134	2387	3763
Sula leucogaster	Brown booby	500	2000	10	375	15	275
Sula sula	Red-footed booby	550	1000	1	714	41	825
Fregata minor	Great frigatebird	50	2400	3	900	0	550
Fregata ariel	Lesser frigatebird	1500	4000	0	16,200	0	3850
Onychoprion fuscatus	Sooty tern	1,000,000	+1,000,000	6000	1,600,000	0	150,000
Onychoprion lunatus	Grey-backed tern	6	1100	25	2000	0	2000
Anous stolidus	Brown noddy	1	10,000	1000	3600	50	1000
Procelsterna cerulea	Blue noddy	1	650	0	26	0	11
Gygis alba	White tern	12	11	0	38	2	50
Nesofregetta fuliginosa	Polynesian storm-petrel	1*	3	0	0	1	0
Puffinus nativitatis	Christmas shearwater	0	20	0	0	0	0
Puffinus bailloni	Tropical shearwater	0	20	0	0	0	0
Puffinus pacificus	Wedge-tailed shearwater	100	41	0	10	0	1*

*Birds found dead
Sources: Clapp and Sibley 1965; Forsell and Berendzen 1986; Sibley and Clapp 1965; Skaggs 1994; US Fish and Wildlife Service 2007

Masked boobies (*Sula dactylatra*) are now widely scattered over Jarvis. The 2004 estimate of 5000 includes several 'clubs' or groups of roosting birds of 1000 or more individuals and represents one of the largest colonies in the world. In 1977, Forsell found hundreds of masked booby carcasses scattered about Jarvis Island. Most were adults, as indicated by the >50 USFWS bands found on the remains. Band recoveries at the 'club' on Jarvis included some from non-breeding birds from Howland and Baker Islands. Cats were observed hunting in groups of up to 20 individuals, killing adult masked boobies (R. Clapp pers. comm.).

In 1982, cats also preyed heavily on sooty terns (*Onychoprion fuscatus*), but by 2004 several hundred thousand were estimated to be in flight over Jarvis. Other visitors have recently estimated numbers of sooty terns there at more than one million individuals (USFWS 2007).

Grey-backed terns (*O. lunata*) were seen occasionally on Jarvis Island when cats were still present. In 1986, biologists found the first grey-backed tern breeding colony of 18 pairs. In 1990, about 50 pairs bred. In 1996, about 100 nests were found at all stages, and by 2004, several hundred birds were breeding.

Brown noddies (*Anous stolidus*) increased from two in 1982 to more than 300 birds in 1986 and by 2004 several thousand were widely scattered over Jarvis Island. Blue noddies (*Procelsterna cerulea*) have also dramatically increased. In 1990, a colony of 36 birds were counted with 11 nests. In 1996, 100 birds were counted. In 2004, we estimated 650, with 274 birds counted at one site.

Procellariiformes were the last seabirds to recover. In 1992 tropical shearwater (*Puffinus bailloni dichrous*, formerly Audubon's, Austin *et al.* 2004) were found nesting. There were no previous records of this species at Jarvis although they nest at Kiritimati. Polynesian storm-petrels, which also nest at Kiritimati, had not been reported alive from Jarvis Island since the 1930s (Bryan 1974) until 2002, when three were seen under coral slabs on the beach crest. None were seen during visits in 2004, 2006 and 2010.

Howland and Baker Islands

The first bird survey of Howland, by the Whippoorwill Expedition in September 1924, recorded 11 breeding species. Expedition members were unable to land on Baker Island, but made estimates from their vessel.

Fig. 2 Pacific crabgrass (*Digitaria pacifica*) covers Howland Island in 1988.

Military activities on Baker Island during World War II eliminated nesting seabirds and by the 1960s, only a few brown noddies were nesting on a small islet in a man-made lagoon, inaccessible to cats. In July 1964, when the cat population had been reduced from 30+ to 4, blue-faced boobies (200 birds, 10 nests), red-tailed tropicbirds (10 birds, 1 nest) and grey-backed terns (3 nests) were nesting, in addition to the noddy terns in the small lagoon (POBSP 1964). In the 1930s, the Pacific crabgrass (*Digitaria pacifica*) was extensive on Baker Island, but not on Howland Island (E. Bryan pers. comm.), suggesting that Pacific rats kept the grass from establishing there until they were extirpated by cats. By the end of 1960s, the rats had been eliminated, the cats had died out, and the aggressive grass was greatly reduced during military operations, but house mice remained (Table 1). A heavy stand of *Digitaria* covered more than half of Howland Island in 1988 (Fig. 2) and this may have played a role in driving nesting seabirds to the more open Baker Island.

From 1942 to the late 1960s most seabird nesting was on Howland Island. In this period rats had been eliminated from both islands. Cats were eradicated from Baker Island by about 1970, but remained on Howland Island. By 1975, most of the nesting seabird species had moved to Baker Island, with the exception of a few thousand frigatebirds that completed their move by 1978 and the last sooty terns moved to Baker by the early 1980s (Table 2). Through the early 1980s the only birds nesting on Howland Island were those that could withstand cat predation. Red-tailed tropicbirds (*Phaethon rubricauda*) nested under coral slabs on the beaches, giving them some protection, but in 1986 cats were preying on some tropicbirds, as feathers were found in a cat stomach and two dead adults, believed to have been killed by cats, were found. Some red-footed boobies (*S. sula*) and frigatebirds no longer nested, but roosted in the kou trees. Several thousand adult masked boobies and a few brown boobies (*S. leucogaster*) that nested in the open on the ground were probably able to avoid predation due to their size and the low numbers of cats. Although there were probably <20 cats present, they seemed to have prevented frigatebirds and terns from re-colonising the island.

In spring 1986, Berendzen and Forsell (1986) removed the remaining 17 cats from Howland Island. In 1988, two years after the cats were removed, the chronology of nesting masked boobies was similar between Howland and Baker Islands. In 1986, similar numbers of boobies were on territories and eggs on both islands, but there were significantly fewer nests with young. Apparently, the boobies are able to protect their eggs and small chicks from cats, but when both adults begin to forage leaving the young unattended, these larger chicks are vulnerable to the cats. This was reflected in the stomach contents of 16 cats examined, of which three had the remains of young boobies.

White terns (*Gygis alba*) returned to Howland by 1992, and brown noddies, grey-backed, and sooty terns by 1998. Red-footed boobies and great frigatebirds (*F. minor)* returned by 1998, and lesser frigatebirds by 2002. A small colony of wedge-tailed shearwaters (*Puffinus pacificus*) was found on Baker Island in 1986 and blue noddies were first found on Baker and Howland Islands in 1993. Numbers of shorebirds rose quickly after cat removal, but surveys in April or later are not a good measure of shorebird abundance as birds migrate back to their Arctic breeding grounds.

The recovery of red-footed booby and great frigatebird numbers on Baker Island was hampered by enormous amounts of debris, primarily thousands of old rusting 55

US gallon (200 L) drums. Some roosting birds, or young on nests built on the rusted tops of the drums, fell in and starved. These hazards were mitigated by turning barrels over so birds could escape, and oil and tar were burned by the U.S. Army Corps of Engineers in 1986. By 1992, most of the debris and entrapment hazards had been mitigated, so it is expected that numbers of great frigatebird and red-footed boobies will grow faster than in the past.

DISCUSSION

Cat predation affected the three islands in different ways. Cats probably extirpated Norway rats from Baker and Jarvis Islands, and Pacific rats from Howland Island (King 1973), but house mice still persist on Baker and Jarvis Islands. They are more resistant to drought than rats, surviving on moisture in insects and condensation (B. Bell pers. comm.).

On Jarvis Island the presence of hundreds of cats eliminated Procellariiformes, shorebirds, and terns, with the exception of sooty terns which still numbered in the hundreds of thousands and were able to sustain a high level of predation at this site (Rauzon 1985). Terns appeared to be a preferred food of cats on all islands (Kirkpatrick and Rauzon 1986). The rapid response of both terns and shorebirds to the removal of cats indicates the impact cats had on the smallest birds. Observations from Howland Island, where less than 20 cats ate chicks and occasional eggs, indicates that masked boobies and tropicbirds can withstand cat predation for many years. Here, a few cats, combined with heavy vegetation, provided enough disturbance to cause the more vulnerable birds to move to Baker Island. Cats then preyed on the remaining boobies and tropicbirds enough to reduce their reproduction. On Baker Island the cats prevented the birds from recolonising, but once cats were eliminated, the colony grew quickly, likely moving from Howland Island. Band recoveries on Jarvis Island show that cats there could have affected birds from a large area, and that the eradication of cats on Jarvis may have contributed to the recovery of masked boobies on other islands (Clapp 1967).

Unlike Jarvis Island, which is separated from the nearest seabird colony, Kiritimati (Christmas) Island, by about 184 nm, Howland and Baker Islands are only 36 nm apart and could be considered a colony complex. Birds have suffered extreme perturbations by man over the past 150 years, but these have often affected only one island at a time, allowing most birds to move back and forth between the islands depending on the severity and type of disturbance on a particular island. When cats were removed from Howland, the birds returned to breed there from Baker (Fig. 3).

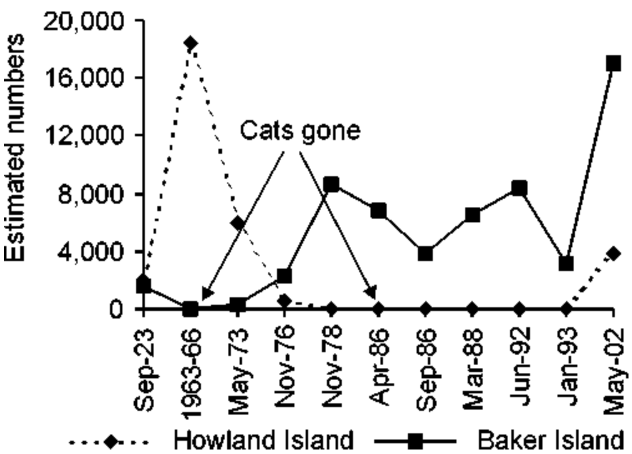

Fig. 3 Combined abundance of great and lesser frigatebirds on Baker (solid line and squares) and Howland (dashed line and diamonds) Islands, before and after cat eradication.

Even though the predation-free period has been longer for Baker than Jarvis Island, fewer new species have recolonised Baker. One reason for this may be the great distance to other colonies that could serve as a source and the condition of those colonies. Another reason is exemplified in the extirpation of Procellariiformes; their high degree of natal philopatry and relative rarity makes them slow to repopulate former colonies. Phoenix petrels (*Pterodroma alba*) were reported from Howland Island in the 1960s, but they did not breed there; and the nearest colony is McKean Island, 352 nm to the southeast. Likewise, Phoenix petrels are expected to visit Jarvis since they also nest at Kiritimati, but unlike Howland or Baker, none are reported in any of the literature reviewed. In 2004, a short-term experiment to attract them with audio recordings failed. Although Polynesian storm petrels nest at Kiritimati, they took 20 years to recolonise Jarvis (where mice may still be a predator) and they still have not been recorded from Baker or Howland, where they were last seen in 1938 (Munro 1944). This may be due to the fact that foraging areas are to the east of Jarvis and Kiritimati, and they are most abundant south of the equator to about 8º S, to the northern edge of the South Equatorial Current, and east to about Marquesas (140º W) (L. Spear pers. comm.).

The 1982-1983 El Niño was the strongest on record and resulted in a severe weakening of trade winds across the Pacific and a significant slackening of the EUC (Firing *et al.* 1983). This effectively caused a complete halt to both regional and local upwelling and resulted in a substantial warming of surface temperatures at each of the islands. With no upwelling, the seabird productivity crashed at Kiritimati. Schreiber and Schreiber (1984) reported that the highest seabird mortality in a 13 year study occurred during the 1982-83 El Niño: "no young fledged during 1982 as they were left to starve to death in their nests by deserting adults." These same oceanographic conditions probably aided the Jarvis eradication in 1982 by stressing cats with low food supplies. Bird populations reached a historic minimum at this time, and seabird recovery began with the cessation of cat predation and the transition to a more productive oceanographic regime.

In contrast to an El Niño phase, the La Niña phase enhances Trade winds and the EUC, and therefore the productivity near these islands (Gove *et. al* 2006). The numbers of blue noddies seen in 2004 are a reflection of the historic strength of the upwelling at Jarvis. These zooplanktivorous, neuston-feeders are more abundant at Jarvis than any other colony. Cats were the apex predators of this marine-based trophic system, which masked the role that upwelling played in the recovery of blue noddies. This recovery may not be only local colony reproduction, but could also reflect immigration from Kiritimati, where this vulnerable tern nests on a few cat and rat-free islets. Kiritimati seabirds never recovered from the 1982-83 El Niño and subsequent human disturbances, and Jarvis Island has become the most significant seabird colony in the central Pacific, as King (1973) predicted it would with cat eradication.

Rats and guano mining destroyed the seabird colonies before ornithologists were able to record the immense populations and diversity of seabirds that this oceanographic and geographic confluence created. After a century of destruction by humans and their commensals, the ecosystems began to recover with the eradication of rats by feral cats. The 25-year effort to control and eradicate cats has allowed almost complete recovery of the seabird biodiversity, if not the numbers, and the islands full status as 'wildlife refuges' has almost been achieved. Mice remain on Baker and Jarvis Islands, and while no predation on seabirds has been observed, their eradication would restore the islands to predator/grazer-free aboriginal conditions.

Continued recovery of these guano island ecosystems is assured with the 2009 protection of their surrounding waters. Most of the recovery parameters needed to reconstitute a guano island ecosystem are in place. However, climatic and oceanographic conditions will ultimately determine if these vulnerable atolls can ever reclaim their immense seabird populations.

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Rodent eradications on Mexican islands: advances and challenges

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Abstract In Mexico, attempts had been made to eradicate rodents from nine islands by 2009, and eight of these were successful. Methods evolved from bait stations on small islands to aerial bait applications on islands that were larger and more complex. Six islands (5 to 82 ha) were treated with bait stations from 1995 to 2002. All of these attempts were successful except for Isabel Island (the largest). Three islands (17, 82 and 267 ha), including Isabel for the second time, were treated with the high-tech aerial broadcast technique between 2007 and 2009, which was the first time this technique was used in Latin America. Post-eradication monitoring has confirmed rodent absence and ecosystem recovery, which includes re-colonisation of seabirds and population increases in reptiles and seabirds. The experience and trust gained have made planning and funding possible for additional projects on bigger islands. Planning, permitting, funding, research, and execution have progressed following a focused, long term collaborative approach with multiple partners. About 30 Mexican islands still have one or two species of invasive rodents. New challenges include bigger islands (e.g., Guadalupe; 24,171 ha), tropical islands (e.g., Banco Chinchorro Atoll; 580 ha), and islands with endemic mammals including rodents (e.g., María Madre; 14,388 ha).

Keywords: Invasive rodents, *Rattus rattus*, *Mus musculus*, Mexican islands, eradication, bait stations, aerial broadcast, ecosystem recovery

INTRODUCTION

Rodents (*Rattus* spp., *Mus musculus*) are among the most harmful and widespread invasive species (Towns *et al.* 2006; Angel *et al.* 2009; Drake and Hunt 2009). They are responsible for the extinction of numerous species of terrestrial vertebrates (Harris 2009), suppress thousands of populations of seabirds (Jones *et al.* 2008), and have significant socioeconomic impacts (Reaser *et al*. 2007). On islands, invasive species such as rodents may establish more easily due to low species diversity and the presence of empty ecological niches (Pino *et al.* 2008).

During the last four decades, eradication techniques developed and tested in New Zealand (Veitch and Clout 2002; Towns and Broome 2003) have been applied successfully on hundreds of islands around the world (Howald *et al.* 2007). There are now numerous examples where eradicating rodents from islands has proved to be an effective way of facilitating the restoration of insular native communities, even in cases where active management beyond rodent eradication is needed (Mulder *et al.* 2009).

Mexico stands out on the American continent because of the high levels of success with the eradication of invasive species from insular ecosystems (Aguirre-Muñoz *et al.* 2008). Applied restoration projects started in 1995 with a couple of rat and cat eradications on small islands, and reached a total of 49 populations eradicated on 30 islands by 2010 (Aguirre-Muñoz *et al.* 2011). The invasive rodent eradications fall into two distinct periods. From 1995 to 2002 six small, dry islands were treated by hand laying bait in bait stations. Two projects were led by researchers of the Universidad Nacional Autónoma de México (UNAM); the rest by Grupo de Ecología y Conservación de Islas-Mexico (GECI) in conjunction with Island Conservation-USA (IC). The second period, which started in 2003, has involved three larger and more complex islands, including one in the tropics. These three were treated with aerially broadcast bait and three more are scheduled for treatment. The projects are part of a long term strategy led by GECI, with the support of key partners. Here we summarise the evolution of these Mexican achievements, and outline plans for the future.

ERADICATION METHODS AND RESULTS

Rodent eradications are particularly difficult (Towns *et al.* 2006). In Mexico, six islands were treated with bait stations: Rasa, San Roque, San Jorge (3 islands) and Isabel. Subsequently, three islands: Farallón de San Ignacio, San Pedro Mártir and again Isabel were treated by aerial broadcast of bait (Fig. 1). These islands were chosen based on the vulnerability of native species to rat predation, on cost and logistical feasibility (in terms of size, topography and native species), and on the level of reinvasion risk. Collectively they are important breeding areas for 19 species of seabirds, 12 species of reptiles, and one species of bat (Table 1). The way that these techniques were applied is described below and additional details are in Table 2.

Fig. 1 Location of Mexican islands were rodent eradications have taken place up to 2009.

Pages 350-355 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

Table 1 Breeding species of vertebrates on Mexican islands where rodent eradications have taken place up to 2009.

Species and common name	Rasa	San Roque	San Jorge	Farallón de San Ignacio	San Pedro Mártir	Isabel
Reptiles						
Aspidoscelis costata Western Mexican whiptail lizard						X
Aspidoscelis martyris San Pedro Mártir whiptail lizard					X	
Aspidoscelis tigris Western whiptail lizard				X		
Crotalus atrox Western diamondback rattlesnake					X	
Ctenosaura pectinata Spiny-tailed iguana						X
Lampropeltis getula nigritus Common kingsnake					X	
Lampropeltis triangulum Milk snake						X
Phyllodactylus homolepidurus Sonoran leaf-toed gecko				X		
Sceloporus clarkii Clark's spiny lizard						X
Urosaurus ornatus Tree lizard				X		
Uta palmeri San Pedro Mártir side-blotched lizard					X	
Uta stansburiana Side-blotched lizard	X	X				
Seabirds						
Anous stolidus Brown noddy						X
Falco peregrinus Peregrine falcon	X	X		X	X	
Fregata magnificens Magnificent frigatebird						X
Larus heermanni Heermann's gull	X	X	X	X	X	X
Larus livens Yellow-footed gull	X				X	
Larus occidentalis Western gull		X				
Pelecanus occidentalis Brown pelican		X			X	X
Phaethon aethereus Red-billed tropicbird			X	X	X	X
Phalacrocorax auritus Double-crested cormorant		X	X	X		
Phalacrocorax penicillatus Brandt's cormorant		X			X	
Ptychoramphus aleuticus Cassin's Auklet		X				
Puffinus opisthomelas Black-vented shearwater		E				
Thalasseus elegans Elegant tern	X	E	X			
Onychoprion fuscatus Sooty tern						X
Thalasseus maximus Royal tern	X	E	X			
Sula leucogaster Brown booby			X	X	X	X
Sula nebouxii Blue-footed booby			X	X	X	X
Sula sula Red-footed booby						X
Synthliboramphus craveri Craveri's murrelet	E	X	E		X	
Mammals						
Myotis vivesi Fish-eating bat	X		X	X	X	
Peromyscus maniculatus cineritius North American deermouse		Ex				

X = presently breeding, E = extirpated, Ex = extinct.

Table 2 Successful rodent eradications on Mexican islands up to 2009.

Island	Area (ha)	Species removed	Date of eradication*	Principal method	Ecosystem type	Reference
San Jorge (3 islands)	<25	*Rattus rattus*	2000	Bait stations	Arid	Donlan *et al.* 2003
Farallón de San Ignacio	17	*Rattus rattus*	2007	Aerial broadcast	Arid	Samaniego-Herrera *et al.* 2009
San Roque	35	*Rattus rattus*	1995	Bait stations	Arid	Donlan *et al.* 2000
Rasa	57	*Rattus rattus* *Mus musculus*	1995[1]	Bait stations	Arid	Ramírez-Ruiz and Ceballos-González 1996
Isabel	82	*Rattus rattus*	2009[2]	Aerial broadcast	Subtropical	Samaniego-Herrera *et al.* 2010
San Pedro Mártir	267	*Rattus rattus*	2007	Aerial broadcast	Arid	Samaniego-Herrera *et al.* 2009

*Work conducted by Conservación de Islas except when indicated otherwise.
[1]Project conducted by J. Ramírez-UNAM (Ramírez-Ruiz and Ceballos-González 1996).
[2]First eradication attempt (1995), conducted by C. Rodríguez-UNAM, failed (Rodríguez-Juárez *et al.* 2006).

Bait stations

Islands treated with this technique were three independent initiatives conducted by different institutions. The first two described below had important seabird nesting sites where researchers had established long term monitoring programmes that documented the negative impacts of introduced invasive rodents. The third one marked the beginning of a large scale island restoration program which now includes all Mexican islands.

1. Rasa Island. In 1995, a ship rat (*Rattus rattus*) and house mouse (*Mus musculus*) eradication was led by Jesús Ramírez (deceased) and collaborators of the Instituto de Ecología, UNAM, and Conservación del Territorio Insular Mexicano, A.C. Bait stations on a 25 m grid containing 50 ppm brodifacoum wax blocks were used; the stations remained for one year although consumption ceased after six weeks (Ramírez-Ruiz and Ceballos-González 1996).

2. Isabel Island. In 1995, a ship rat eradication was undertaken by C. Rodríguez and collaborators of the Instituto de Ecología, UNAM. Bait stations containing 50 ppm brodifacoum wax blocks were used; the bait stations were removed after just six weeks even though consumption rates of the baits had not decreased (Rodríguez-Juárez *et al.* 2006).

3. San Roque and San Jorge Islands. Unlike the above projects, eradications on these islands were part of a larger scale strategy of island restoration work. GECI, in conjunction with IC, started applied restoration work on Mexican islands in 1995 eradicating cats (*Felis catus*) and ship rats on San Roque Island. Brodifacoum wax blocks were used to eradicate rats in combination with 100 ppm bromethalin in a gel bait; stations remained for one year (Donlan *et al.* 2000). Later in 2000, ship rats were eradicated from all three San Jorge Islands. Bait stations on the biggest island were on a 25 m grid and contained 50 ppm brodifacoum wax blocks. On east islet diphacinone was used and on the west islet cholecalciferol was used. The bait stations on each island remained in place for one year (Donlan *et al.* 2003).

Aerial broadcast

Following the experience gained by working on small islands and with growing support of funders and partners, GECI then initiated more ambitious projects. Because Farallón de San Ignacio, Isabel and San Pedro Mártir islands are topographically complex, and the last two are medium sized (82 and 267 ha, respectively), a helicopter was used to disperse rodenticide broadly across each island. Although effectively employed elsewhere (e.g., Howald *et al.* 2007), this was the first use of aerial procedures in Latin America. Each rat eradication project included a two year pre-eradication and a two year post-eradication phase. In all cases the ship rat was the target species.

Farallón de San Ignacio and San Pedro Mártir islands were integrated into a single project due to their physical and ecological similarities. Both islands were treated in autumn 2007 using specially designed 25 ppm brodifacoum pellets manufactured by Bell Laboratories, USA (for additional details see Samaniego-Herrera *et al.* 2009).

Isabel Island is the most recent project. Although the previous attempt to eradicate rats from this island using bait stations failed, the aerial broadcast of baits in spring 2009 appears to have been successful. No rats have been detected following almost two years of monitoring. This eradication used the same Bell Laboratories bait described above but with the addition of a biomarker, which allowed monitoring of consumption by target and non target species, especially those scarcely or non present in previous eradications (e.g., iguanas, snakes, land birds; see also Samaniego-Herrera *et al.* 2010).

Additional research and activities

There was limited monitoring of native species on the islands treated with bait stations and the results remain unpublished. Existing information comprises changes in seabird populations on Rasa and San Roque islands (Table 3). Pre and post-eradication monitoring on the islands treated with aerial broadcast (Table 2) included reptiles, seabirds and bats on all islands (Table 3). On Isabel Island

Table 3 Examples of ecological benefits on native populations at Mexican islands after rodent eradications.

Island	Species	Changes recorded after rodent eradication
Farallón de San Ignacio	*Phaethon aethereus* Tropicbird	60% increase in number of nests after two years without rats. Percentages of egg-hatching success and development of juveniles also increased.
	Phyllodactylus homolepidurus Sonoran leaf-toed gecko	Changed from extremely rare to low abundance after two years without rats.
Isabel	*Ctenosaura pectinata* Spiny–tailed iguana	Population abundance increased.
	Onychoprion fuscatus Sooty tern	Nesting again after few years of extirpation.
Rasa	*Larus heermanni* Heerman's gull	Breeding success increased five times.
	Thalasseus elegans Elegant tern	Population (55,000 individuals in 1995) has increased to 200,000.
San Pedro Mártir	*Lampropeltis triangulum* Milk snake	"Reappeared" on the island after two years without rats.
	Synthliboramphus craveri Xantus´s murrelet	Nesting again after decades of extirpation.
San Roque*	*Phalacrocorax penicillatus* Brandt's Cormorant	Both nesting again after years of extirpation. Also several new records of seabirds in recent years.
	Ptychoramphus aleuticus Cassin's auklet	

* The project included both ship rat and cat eradication.
Sources: Velarde *et al*. 2005; Castillo 2009; Samaniego *et al.* in prep; E. Velarde pers. comm.

terrestrial crabs were also monitored. Details of the species involved, methods, and results will be provided elsewhere (Samaniego-Herrera *et al.* in prep).

Several biomarker trials have been associated with planed rodent eradications (Greene and Dilks 2004; Griffiths *et al.* 2008; Parks and Wildlife Service 2009; Wegmann *et al.* 2009). To the best of our knowledge, the 2009 rat eradication on Isabel Island was the first to use bait with a biomarker for the actual toxic bait application. This "large scale experiment" allowed us to test the palatability of baits across a wide range of native and introduced species of invertebrates and vertebrates. The results are part of a larger study, which include other insular ecosystems, so are not reported here.

All projects included environmental education as a tool for both project acceptance by local communities and authorities, and for reinvasion risk management. On seven of the eight islands the risk of reintroduction is low because the islands are not inhabited by humans, have no tourism, or no longer feature activities with a high risk of accidental introduction of invasive rodents (mainly guano mining). The exception is Isabel Island, which is inhabited by fishermen for most of the year and is regularly visited by small groups of students and tourists. Due to the higher risk of reintroduction, the authority with jurisdiction over this natural protected area, which is the Comisión Nacional de Áreas Naturales Protegidas (CONANP), enforces an environmental education campaign with a permanent prevention programme that includes checks of boats and gear for invasive species.

Along with the eradication projects and field monitoring, literature reviews and interviews were conducted to update our database of invasive rodent distribution on Mexican islands. Monitoring included standard trapping of small mammals in different habitat types and seasons when possible. Inputs from authorities, island users, and researchers included formal interviews and informal conversations. Most of the cases revealed from interviews were confirmed in the field, and the rest were backed by credible evidence.

DISCUSSION

Five of the six attempted eradications using bait stations were successful on islands <52 ha (Table 2), but the attempt failed on the largest island, Isabel (82 ha). Hasty implementation without first studying the behaviour and ecology of the target population (as well as potential native competitor species) were identified as the main cause of failure (Rodríguez-Juárez *et al.* 2006). We agree that insufficient planning and research, especially concerning land crab interference, and not the method, was the cause, especially since much bigger islands have successfully been cleared of rats with bait stations (Howald *et al.* 2007).

All three projects using the aerial broadcast of baits were successful. On the most recent one (Isabel Island), the second year of confirmation monitoring is about to be completed. As in New Zealand and elsewhere, the size and complexity of islands favoured a change of methods from bait stations to aerial broadcast procedures. Baits and techniques developed in New Zealand, which in turn were adopted in the USA, facilitated the several technical, logistical and legal aspects involved in these eradications. At the same time it is important to highlight that crucial requirements for aerial procedures, such as helicopters equipped with Differential GPS and permits to import the specific rodenticides required by the method, are difficult to obtain in Mexico. Therefore these will continue to be limiting factors for future projects until a facilitating legal framework for restoration projects is developed. This concern has been underlined in previous publications (Aguirre *et al*. 2005, 2008, 2009) and national forums attended by researchers as well as managers and government authorities (e.g., Encuentro Nacional para la Conservación y el Desarrollo Sustentable de las Islas de México, 2009).

The positive effects observed after rat eradications in Mexico include increased reproductive success and recolonisation of seabirds, as well as increases in the abundance of reptiles (Table 3). On Isabel Island, the eradication of the invasive house sparrow (*Passer domesticus*) was an additional but unplanned benefit; once common around human settlements, no sparrows have been recorded in almost two years of monitoring (Samaniego-Herrera *et al.* unpublished data).

Environmental education and re-invasion prevention programmes, combined with a low to moderate risk of reintroduction, have so far helped to prevent reinvasions by rats. The first projects were completed about 15 years ago and recent field monitoring confirms that the islands are still free of invasive rodents (Samaniego-Herrera *et al.* 2007). Moreover, these eight islands are now free of all invasive mammals as cats were also eradicated from some of them.

There are at least 30 more Mexican islands with either ship rats, house mice, or both species present (Table 4). There are also two invasive species that are native to an adjacent area: *Peromyscus eremicus cedrocensis* which is endemic to Cedros Island, was accidentally introduced to San Benito Oeste Island (50 km east of Cedros) in 2006 (Aguirre *et al.* 2009); *P. fraterculus*, which is native and common on the adjacent mainland, was probably introduced to Santa Catalina Island in the beginning of the 1990s (Álvarez-Castañeda *et al.* 2009). There are no confirmed records of brown rats (*R. norvegicus*) on islands although the species is present on mainland Mexico. Some of the remaining invaded islands are small and rodent eradication should be easily achievable with baits in stations or broadcast by hand. However, several islands are very close to either the mainland or to a larger island with invasive rodents, hence elevating the risk of reinvasion. Eradication must then be evaluated in a cost-benefit perspective, as management requires a metapopulation approach (Russell *et al*. 2009) and expensive prevention considerations must be taken into account. Regarding the islands on which aerial broadcast is the only option to eradicate invasive rodents, size is not the only challenge. Apart from human activities, tropical ecosystems, and the presence of native mammals, including rodents, are the biggest concerns; factors for which there is little experience worldwide (Wegmann 2008; Harris 2009; Varnham 2010). In preparation for future eradication projects we are conducting monitoring and research on topics such as species that indicate ecosystem recovery, the palatability of baits and the risks they pose to native species, and mitigation measures for those species at risk of primary and secondary poisoning.

The successes on all eight islands prove support for the initiative to scale up the rodent eradication programme at a national level. More than ever, rodent eradications in Mexico should constitute an inter-institutional effort and prioritisation analyses need to be developed. Funding must be secure in advance and include pre- and post-eradication studies and environmental education, and bio-security measures need to be applied in a serious and long term approach.

Table 4 Mexican islands with presence of exotic invasive rodents in 2010.

Island	Area (ha)	Species	Ecosystem type	Native mammals?
PACIFIC OCEAN				
Cedros	34,933	*Mus musculus* *Rattus rattus*	Temperate	Yes
Coronado Sur	126	*Mus musculus*	Desert	Yes
Guadalupe	24,171	*Mus musculus*	Temperate	No
Magdalena	27,773	*Mus musculus*	Desert	Yes
San Benito Oeste	364	*Peromyscus eremicus cedrosensis*	Desert	No
Socorro	13,033	*Mus musculus*	Tropical	Yes
GULF OF CALIFORNIA				
Alcatraz (Pelícano)	50	*Mus musculus*	Desert	No
Almagre Chico	10	*Rattus rattus*	Desert	No
Ángel de la Guarda	93,068	*Mus musculus* *Rattus rattus*	Desert	Yes
El Rancho	232	*Mus musculus* *Rattus rattus*	Desert	No
Granito	27	*Rattus rattus*	Desert	Yes
María Madre	14,388	*Rattus rattus*	Tropical	Yes
María Magdalena	6977	*Rattus rattus*	Tropical	Yes
María Cleofas	1963	*Rattus rattus*	Tropical	Yes
Mejía	245	*Mus musculus* *Rattus rattus*	Desert	Yes
Melliza Este	1	*Rattus rattus*	Desert	No
Pájaros	82	*Rattus rattus*	Desert	No
Saliaca	2000	*Mus musculus* *Rattus rattus*	Desert	Yes
San Esteban	3966	*Rattus rattus*	Desert	Yes
San Vicente	14	*Mus musculus*	Desert	No
Santa Catalina	3890	*Peromyscus fraterculus*	Desert	Yes
GULF OF MEXICO AND CARIBBEAN SEA				
Cayo Norte Menor	15	*Rattus rattus*	Tropical	No
Cayo Norte Mayor	29	*Rattus rattus*	Tropical	No
Cayo Centro	537	*Rattus rattus*	Tropical	No
Cozumel	47,000	*Mus musculus* *Rattus rattus*	Tropical	Yes
Holbox	5540	*Rattus rattus*	Tropical	No
Muertos	16	*Mus musculus*	Tropical	No
Mujeres	396	*Mus musculus* *Rattus rattus*	Tropical	No
Pájaros	2	*Mus musculus*	Tropical	No
Pérez	11	*Rattus rattus*	Tropical	No

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Eradicating mammal pests from Pomona and Rona Islands in Lake Manapouri, New Zealand: a focus on rodents

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Abstract Pomona and Rona Islands are situated in Lake Manapouri, Fiordland National Park New Zealand. Since 2006 a community-driven initiative, led by the Pomona Island Charitable Trust, has been removing the introduced pests from the two islands. Rona Island had stoats (*Mustela erminea*) and mice (*Mus musculus*) present, Pomona Island had five pest species to be removed: stoats, red deer (*Cervus elaphus*), possums (*Trichosurus vulpecula*), ship rats (*Rattus rattus*) and mice. Pomona and Rona Islands are 500m and 600m respectively from the mainland. Having removed the stoats from Rona Island and the stoats, deer and over 430 possums from Pomona Island, the Trust undertook an aerial operation to eradicate the rodents from both islands. Following best practice, the aerial operation involved two aerial drops of the pesticide brodifacoum conducted 40 days apart. The paper provides an overview of the eradication techniques for each pest species on the islands, planning for the rodent eradication operation, community consultation, logistics of the aerial operation and post-eradication monitoring to confirm the success of the operation. Biosecurity measures put in place post-rodent eradication are discussed and their effectiveness assessed.

Keywords: Eradication, rats, *Rattus rattus,* mice, *Mus musculus,* brodifacoum, community conservation

INTRODUCTION

Pomona and Rona Islands are located in Lake Manapouri, Fiordland National Park, New Zealand (Fig. 1). Pomona Island, 262 hectares, is the largest island in a lake in New Zealand. Rising 340 m above Lake Manapouri, it is a round-topped granite hill with steep sides separated from the mainland by the 500m wide Hurricane Passage. The island is almost completely forested and has some impressive bluffs. Pomona Island has a variety of habitat types and a rich flora for its size. Introduced predators and browsers have had an impact on the island's biodiversity, particularly native birds. Five mammal pests have been recorded on Pomona: stoats (*Mustela erminea*), ship rats (*Rattus rattus*), possums (*Trichosurus vulpecula*), mice (*Mus musculus*) and red deer (*Cervus elaphus*).

Rona Island (60ha) is the second largest island in Lake Manapouri and is just over 600 m from the mainland. Two pest mammal species have been found there: stoats and mice.

The Pomona Island Charitable Trust (a community-led organisation) was established in 2005 with the vision of restoring Pomona and Rona Islands to a pest-free state and maintaining them as island sanctuaries. The key aims of the Trust are to eradicate all introduced pest species from the islands, to re-introduce, through natural and assisted means, birdlife native to Fiordland and the Southwest New Zealand World Heritage Area, to ensure community involvement in the island restoration project and to provide an accessible location for people to see, hear and learn about the flora and fauna native to Fiordland. The Trust has a management agreement with the Department of Conservation (DOC) to manage the restoration project on the two islands

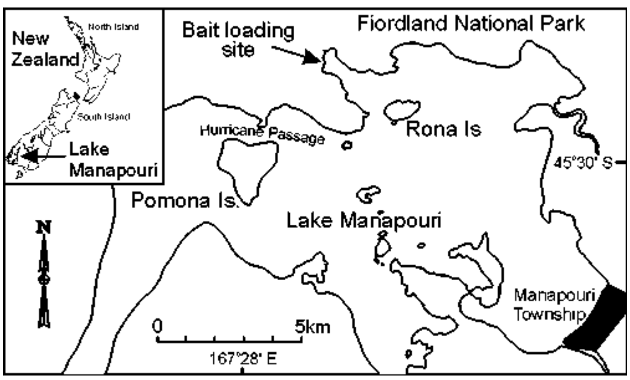

Fig. 1 Location of Pomona and Rona Islands.

MAMMAL ERADICATIONS

Eradication of Stoats, Possums and Deer

A formal pest management plan was commissioned to guide the eradication of pests from the two islands (Brown 2006). The work described in this paper follows this plan with modifications as noted.

Stoats were the first pest to be targeted. A 9.2km network of tracks was cut by volunteers on Pomona Island to service 37 stoat trap sites. Each stoat box contained a double-set Mark IV Fenn trap which was baited with an egg and a piece of meat (Fig. 2). The traps were first set in August 2006 and, up to November 2007, 18 stoats were trapped. In September 2008 the double-set Fenn traps were replaced with single DOC 150 traps and in September 2009 an additional 10 single-set DOC 200 traps were placed along the Hurricane Passage side of the island, the part of the island most vulnerable to re-invasion.

In January 2007 four double-set Mark IV Fenn traps were set out around the coast of Rona Island. Three stoats were caught in these traps, the last one caught in January 2008. In October 2009 a further 14 double-set DOC 150 traps (Fig. 3) were installed across the island and no stoats were caught up to February 2010.

Possums were introduced to Pomona Island in the 1970s by a hunter who wanted his own personal supply of possum fur. The number of possums present on the island was unknown, but estimates from possum hunters were around 200 animals. Possums are known to eat the bait that was to be used for the rodent eradication, so it was deemed important to remove as many as possible from the island prior to the aerial application of brodifacoum. In May 2007 a contractor was employed and used a mix of leg-hold traps and Feratox poison to kill more than 430 possums. There was a concern that, with so many dead possums on Pomona Island, the carcasses might provide an alternative food supply for the rodents. For this reason the contractor and volunteers removed as many carcasses as possible. Where it was not practical to remove them from the island, the contractors created piles of possums

Pages 356-360 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

Fig. 2 A double-set Mark IV Fenn trap baited with an egg and a piece of meat.

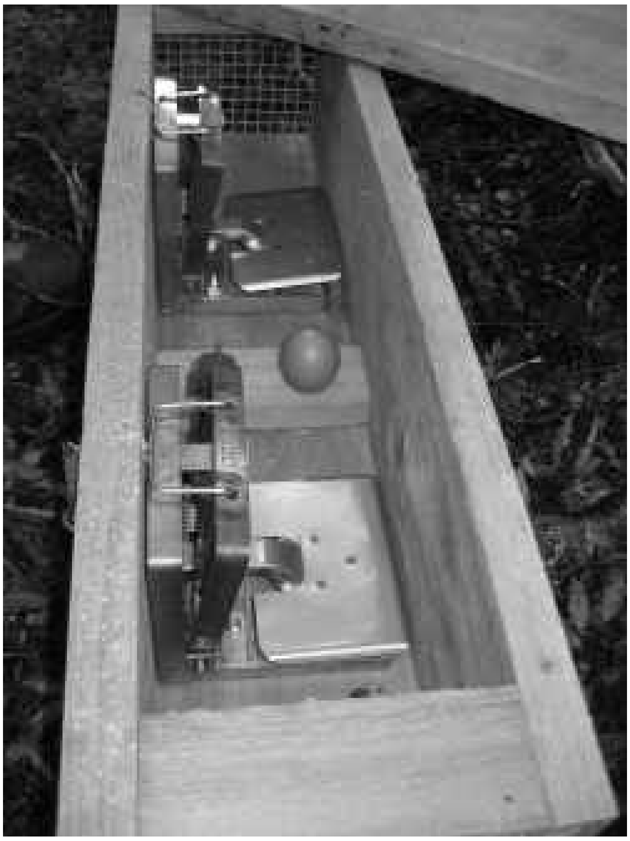

Fig. 3 A double set of DOC 150 traps baited with an egg and a piece of meat.

in identified locations along the track. To ensure that all possums had been eradicated a second possum operation was conducted in June 2008, but no evidence of possums was found.

Deer: Pomona Island is well within the swimming range of red deer. In the past, local hunters have sporadically hunted deer on the island and a pen to trap deer had been constructed there during the 1970s. Given that the deer on the island had been subject to some hunting pressure, it was felt that a professional contractor with dogs would be the best option to remove them. A condition of the resource consent for the aerial poison operation targeting rodents was that all practicable steps be taken to remove deer from the island prior to the rodent eradication programme. During May 2007 a total of five deer were shot on the island and the deer pen was repaired and re-activated. Since then no further evidence of deer has been found on the island.

Eradication of Rodents

Choice of Method

An aerial poison operation using brodifacoum-laced bait was selected as the best method for eradicating rats and mice from Pomona Island and mice from Rona Island. The reasons for this were that the cost of ground-based control would be very high and the steep nature of the terrain, especially on Pomona Island, would mean that complete coverage could not be guaranteed. As a charitable organisation, the Trust had limited funds and would not have had the ability, financially or in terms of volunteer resource available, to conduct on-going ground control work. A one-off aerial operation, therefore, represented the most cost effective approach to rodent eradication. Brodifacoum has been used to successfully eradicate rodents from a number of islands and at the time of the operation the proposed method was also the 'best practice' for eradication of rats and mice from islands (e.g., Clapperton 2006; Clout and Russell 2006; Torr 2002; Veitch 2002a; Veitch 2002b; Veitch 2002c; Empson and Miskelly 1999).

Operational Planning

The pest management plan (Brown 2006) prescribed two aerial applications of bait, spaced a minimum of seven days apart. It recommended sowing bait at a nominal rate of 8kg/hectare with two extra swaths along the entire coastline of both islands for the first application and a second application sown at a nominal rate of 4.5kg/hectare with two additional coastal swaths. A review by the Department of Conservation's Island Eradication Advisory Group (IEAG) recommended increasing the sowing rate to 8kg/hectare for both drops, with two additional coastal runs and increasing the minimum time between the drops to 10 days. Following an inspection of the islands by the Chief Pilot for the operation, Peter Garden, he recommended additional bait be sown on the steeper flanks of Pomona to ensure good coverage. Therefore, in addition to the two coastal runs, some of the bluffs on the island received as many as six additional coastal runs.

The aerial spread of poison required a resource consent from Environment Southland. An Assessment of Environmental Effects provided an overview of the proposed operation, a description of the treatment area, a discussion of alternative rodent eradication options available, the environmental effects of using brodifacoum and a set of proposed consent conditions (Willans 2007). Resource consent was received in May 2007. A total of 7.1 tonnes of Pestoff 20R cereal bait containing 20ppm (0.02g/kg) of brodifacoum was ordered. This included a 10% contingency amount to allow for any unforeseen mishaps with the bait applications or the need to re-treat any gaps in bait spread.

The poison drops were planned for the winter of 2007. Winter was judged as the most appropriate time for the operation, as food supplies for the rodents would be at their lowest thus increasing the chance they would eat the bait. It was also judged desirable to ensure that as many of the possums and deer on the islands as possible had been eliminated prior to the first drop so that the competition for the bait would be reduced. Following each aerial operation volunteers laid additional bait by hand around the piles of possum carcasses on Pomona.

Community Involvement

A Social Impact Assessment (see Cosslett *et al.* 2004) for the operation was undertaken to identify key interested parties in the local community and to ascertain their views on the Trust's planned restoration of Pomona and Rona Islands and, in particular, the planned method of rodent eradication (Shaw 2006). Members of the community were overwhelmingly positive to the Trust's plans to restore Pomona Island. The vast majority of respondents thought that this was a great project which would benefit the local communities of Manapouri and Te Anau. Strong support was found for the Trust's plans to eradicate all pests from the island and re-introduce native bird species. There was widespread support for the community-driven initiative with high numbers of individuals volunteering their time to the project. Support for the project from the Department of Conservation was considered by members of the community to be important. A strong relationship with the Department has developed as the restoration project has progressed.

The main concern raised at a public meeting related to the poison that the Trust planned to use. When informed that brodifacoum (the active ingredient in the product Talon which is freely available for household use in New Zealand) was the poison recommended in the pest management plan, concerns seemed to be allayed. A small number of individuals raised the issue of alternatives to an aerial poison drop and questioned whether it was possible to eradicate the rodents using hand-laid bait. Research evidence suggests that the spacing of bait would have to be very close. The manufacturer's recommendation for Talon for mouse control is that bait should be no more than 3 m apart (Clapperton 2006). The cost of adopting such an approach would be high and would be impractical due to the nature of the terrain on both islands. Once this was explained, the individuals expressing their concern seemed to accept the rationale for an aerial poison drop.

The Trust kept the local community informed about the operation. Public meetings were held and an information sheet prepared, distributed and put on the Trust's website. Objection to the operation from deer hunters resulted in a condition placed on the Trust to take all practical steps to eliminate deer from Pomona Island prior to the aerial operation.

As a community-led restoration project there was strong volunteer involvement in the actual rodent eradication programme. The Project Manager, Operations Manager and the Chief Pilot all donated their time and volunteers, under the supervision of Department of Conservation staff, loaded the bait into the spreader bucket slung beneath the helicopter. Volunteers did all the post-operation ground checks for bait coverage and the condition of bait on the ground. Funding for the aerial operation came from community sources, with NZ$40,000 being donated by an anonymous benefactor and the remaining NZ$14,195 coming from the Community Trust of Southland. Weather forecasting for the operation was also provided by a local contractor to the Trust at no charge.

Aerial Operation

Bait was sown using a Bell Jet Ranger helicopter with an under-slung spreader bucket with an effective swath width of 80 m. A Differential Global Positioning System (DGPS) with a fixed base station was used to guide the helicopter whilst sowing bait. Bait was sown at 4kg/ha with a 50% overlap of swaths giving the target coverage of 8kg/ha. The first aerial operation took place on 8 July 2007. The DGPS base station was installed on the mainland close to Manapouri, giving coverage of both islands. Volunteer bait loaders and the bulk bait were flown to the loading site on the mainland adjacent to the islands (Fig. 1) where the bait was loaded into the spreader bucket. An experienced GIS expert from the Department of Conservation, capable of downloading and interpreting the logged flight data from the helicopter's DGPS, joined the crew at the loading site. Bait was spread first on Rona Island and a printout from the DGPS unit assessed to ensure coverage was complete. Bait was then spread on Pomona Island. Data was downloaded from the DPGS and coverage assessed before all of the equipment and volunteers packed up for the day. The loading site was cleared to ensure that no pellets remained on the ground. Poison warning signs were put in place by volunteers on the islands and at all boat launch sites on Lake Manapouri.

The weather for the first drop was perfect, with freezing conditions on the ground. There was no significant rainfall for 16 days following the first drop. An inspection of the bait on the ground a week after the aerial operation showed good coverage had been achieved and the cold, crisp conditions meant that the bait was still in almost pristine condition. The second aerial operation was therefore delayed until 18 August 2007. The second operation was conducted in an identical fashion to the first. There was no significant rainfall for eight days following the second drop and a ground inspection found good coverage on both Pomona and Rona Islands. Bait was still visible on the islands three months after the second aerial operation. Three dead chaffinches (*Fringilla coelebs*) were found on Pomona Island a month after the first aerial operation.

Possible Re-invasion

Rodent motels and bait stations were placed on both islands to detect possible survivors or a re-invasion (four motels and 12 bait stations on Pomona and one motel and

Fig. 4 A rodent motel.

four bait stations on Rona). The motels (Fig. 4) contain two mouse traps, two rat traps and two poison bait blocks with an area in the middle for rodents to sleep.

In June 2009, a single mouse was found in a trap inside a rodent motel on Rona Island. In response to this, the Trust, with the assistance of the Department of Conservation, placed over 50 temporary mouse traps on Rona Island around the site of the mouse find, and in other potentially vulnerable sites. In October 2009, a network of 30 mouse traps was placed permanently on the island in areas that are most vulnerable to re-invasion. In addition, a mouse trap was permanently sited inside each of the 18 stoat traps on Rona, giving good coverage across the island. These traps are checked monthly and no further evidence of mice has been found on Rona Island.

In July 2009 a single mouse was also found in a trap inside a rodent motel on Pomona Island. Judging by its condition, it had been in the trap for a while so may have appeared on Pomona around the same time as the mouse found on Rona Island. Fifty temporary mouse traps were placed around all potential landing sites on the island. Again these traps have now been replaced with permanent mouse traps in vulnerable locations and mouse traps have been placed inside each of the 47 stoat traps boxes on the island. No further evidence of mice has been found on Pomona.

Both mice were caught in traps located close to preferred boat landing sites on each island. In spite of intensive trapping in the location of both finds, no further evidence of mice has been found on either island. It is likely that the single mice found on each island were the result of re-invasions rather than remnant populations on either island. The most likely source of re-invasion is from a boat.

POST ERADICATION MONITORING AND BIOSECURITY

Pomona and Rona are Open Sanctuary Islands and are accessible to the public. Anyone with their own boat can visit either island at any time. Biosecurity is an important issue and is being handled in three ways: i) through on-going monitoring on both islands, ii) through the installation of trap lines on the adjacent mainland and iii) through public education.

Monitoring

There are 47 traps capable of catching both stoats and rats permanently in place on both islands. A network of 16 bait stations and rodent motels have been placed on Pomona Island and five on Rona Island. The 12 bait stations contain brodifacoum poison bait and the four rodent motels contain rat traps, mouse traps, chew sticks and poison bait. Chew sticks containing peanut butter have also been placed alongside each stoat trap location on both islands and along some of the tracks around Pomona Island. The chew sticks may identify the presence of rats, mice and possums. No evidence of animals chewing the chew sticks has been found between August 2007 and February 2010. One stoat has been caught on each island between the aerial poison operation in July 2007 and February 2010. The traps are checked and bait replaced bimonthly. Chew sticks are replaced quarterly.

Mainland Trap Lines

A network of stoat and rat traps has been established on the mainland adjacent to the two islands. The rationale is to reduce the risk of re-invasion of either island by stoats. In October 2006 24 double set DOC 150 traps were set out along the coast opposite Pomona Island. In September 2009 an additional 48 single set DOC 200 traps were placed on the peninsula and along the coastline opposite Pomona. This was done in response to a moderate beech mast in Fiordland, which would be expected to lead to an increase in the numbers of stoats and to reduce the risk of stoats swimming across to Pomona. Between October 2006 and February 2010 a total of 73 stoats, 156 rats and 14 mice were caught in these traps.

In order to protect Rona Island from a potential stoat re-invasion, two mainland trap lines were established at the closest points to the island. Ten double set DOC 150 traps on the mainland, approximately 980 m to the north of Rona, caught nine stoats, two weasels, 24 rats and four mice between October 2008 and February 2010. To the west of Rona and only 600 m away, a network of 11 double set DOC 150 traps and 23 single set DOC 200 traps caught six stoats, 69 rats and 12 mice between October 2008 and February 2010. The mainland traps are checked and freshly baited every two months, with the frequency increased to monthly following a beech mast event.

Education

Since completing the eradication of pests from Pomona and Rona Islands, the Trust has turned its attention to educating the local community on the need to keep the islands free of introduced animal pests, especially rats and mice. The Trust has produced a quarantine brochure aimed at users of the lake. These are available at all boat launch sites on Lake Manapouri and encourage users of the lake to help protect Pomona and Rona by ensuring that they do not accidentally re-introduce rodents to the islands. Boat owners are encouraged to have rodent bait stations or traps on their boats to help minimise the risk. Local boat clubs have been informed of the islands' pest-free status and are asked to encourage their members to adopt the necessary precautions to keep them pest-free. All volunteers and commercial boat operators that visit the two islands to work on the restoration project have been provided with a bait station and rodent traps for their boats. Permanent signs at key landing sites on the islands inform the public of their pest-free status and provide a reminder of the checks that individuals should undertake before setting foot ashore.

RESTORING POMONA AND RONA ISLANDS

No rats have been seen or trapped on either island since the second aerial poison operation in August 2007. Eradicating pests on both Pomona and Rona Islands simultaneously has proven to be a cost effective approach to the islands' restoration and having two island sanctuaries close together acts as an insurance policy for species native to Fiordland. In the unlikely event that one of the islands suffered a re-invasion of pests, the flora and fauna on the other is still safe and can be used to re-populate the other island if needed.

Five minute bird counts on both Pomona and Rona Islands show that the numbers of birds have increased significantly as a result of the eradication of pests from both islands. Baseline bird counts were undertaken prior to the pest eradication programme (Porter and MacTavish, 2006). Pomona Island has seen an increase of 103% in the number of birds recorded and Rona Island an increase of 50% following the eradication (Fig. 5). The smaller increase in the number of birds recorded on Rona Island since pests were eradicated could be a consequence of the fact that this island had no rats or possums prior to the eradication, so may have been less affected than Pomona.

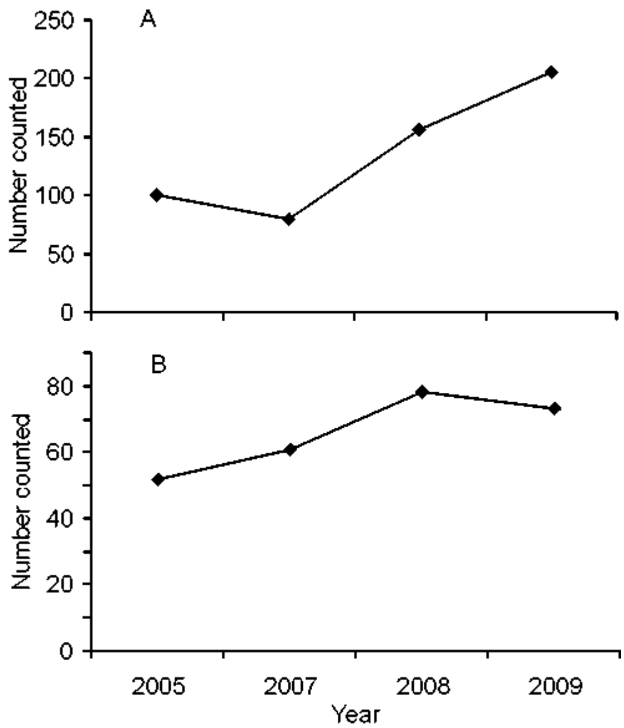

Fig. 5 Bird count data from Pomona Island (A) and Rona Island (B).

The Pomona Island Charitable Trust has now shifted its attention away from pest eradication to maintaining the islands as pest-free sanctuaries and to restoring Pomona and Rona back to their former glory. A restoration plan has been prepared (Shaw and Whitehead 2008) and, in February 2009, the first of many planned species translocations took place with the transfer of South Island robins (*Petroica australis*) to the two islands. Translocations planned for the future include mohua (*Mohouoa achrocephala*), saddleback (*Philesturnus carumculatus*) and kiwi (*Apteryx australis*).

ADDENDUM

In March and May 2010 single mice were trapped on Pomona Island. In addition to the 92 mouse traps, a network of 84 tracking tunnels was placed on the island. In June 2010 six mice were trapped and since August 2010 a further 78 mice have been trapped in locations across the whole island. Mouse tracks have been found in 80% of the tracking tunnels. With the assistance of the IEAG, DNA testing of the island mice versus a sample of mainland mice will be undertaken.

In March 2010 a singe mouse was trapped on Rona Island. In addition to the 60 mouse traps, 16 tracking tunnels were placed on the island. No mice have been trapped and no evidence of mice has been found in the tracking tunnels.

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Thanks to the Department of Conservation for the technical support provided throughout the eradication programmes and also for trusting us to get on and do the job properly.

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Eradication of the house crow from Socotra Island, Yemen

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Abstract The house crow (*Corvus splendens*) is one the world's most invasive bird species, affecting more than 25 nations throughout the Indian Ocean, Arabian Peninsula and South East Asia. It can create problems for the natural biodiversity of regions, as well as impacting upon human health, tourism, infrastructure, and general development. The first house crows arrived on Socotra Island, Republic of Yemen, in 1995/96, having been transported unintentionally from mainland Yemen by ship. Socotra Island is a UNESCO world heritage site and its ecosystem includes a large number of endemic species. For more than a decade, ongoing management on the island by bounty payments kept numbers of the breeding population low, but did not remove it completely. The population of the last 13 birds was eradicated by shooting in April 2009, Management of other populations of this invasive species is recommended.

Keywords: *Corvus splendens*, chick collection, shooting, monitoring, regional cooperation, small populations, spread, bounty

INTRODUCTION

The house crow (*Corvus splendens*) is native to India and parts of its neighbouring countries where it is closely associated with people (Ali 2002) and has some negative impacts on their communities. However, the effects of house crows are so significant in the 25 or more countries throughout Africa, the Middle East and South East Asia where the bird has been introduced (Ryall 1994, 1995, 2002), it is now regarded as one of the world's most invasive bird species (GISD 2010). Introduced house crows continue to spread across the region of their introduction (Nyari 2006) with negative effects on agriculture, tourism, human health, traffic, transport, and biodiversity (Ryall 1992b). House crows eat crops and damage orchards (Dhindsa *et al*. 1991; Feare and Watson 1990); disturb tourists and local citizens with their loud calls, as well as their heavy defecation and aggressive attacks when attempting to steal food (GISD 2010); transmit pathogens, which affect people and domestic animals (Al-Sallami 1991; Cooper 1996; Roy 1998); and also pose a bird strike risk to aeroplanes (Ryall 1992b). The crows are also responsible for the reduction or severe depletion of small reptiles and amphibians, birds and mammals, insects, fish and domestic animals (GISD 2010). Lack of data allows no quantification of such losses and disturbances. However, in the areas that are newly colonised by this bird species, the impact is believed to be high. In most of the affected countries, no control projects against the house crow are undertaken.

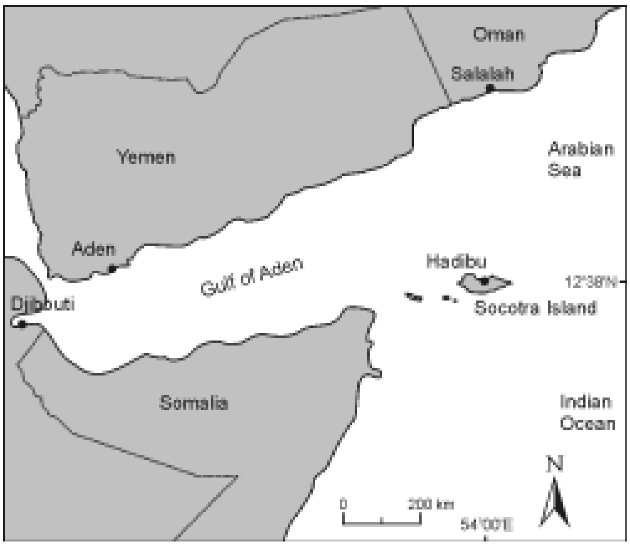

Fig. 1 Location of the island of Socotra, Republic of Yemen, and other locations mentioned in the text.

This paper records the arrival, establishment, and measures used to control, and subsequently eradicate the house crow on Socotra Island. This work was managed by staff of the Socotra Environmental protection Agency (SEPA). There was no funding or action for detailed pre-eradication research or planning. Biosecurity measures for possible new house crow arrivals are not considered in this paper.

PROJECT SITE

Socotra Island (3500 km^2), in the Republic of Yemen, is 380 km off its coast and 150 km from the horn of Africa (Fig. 1). The human population of 43,000 is not dense due to the remote location and desert environment.

The island has 65 % endemism of the approximately 900 species of plants and up to 90 % endemism of insects and reptiles (Unpubl. SEPA data). Socotra became a UNESCO World Heritage Site in 2007, which demonstrates the value of the island for the region's biodiversity but also its value for tourism. The island's terrestrial environment is threatened by uncontrolled development and its surrounding waters by illegal fishing, but invasive animals were not considered a threat for many years.

The house crow arrived on Socotra Island in 1995 or 1996 (Table 1), when one pair was thought to have travelled on a ship and then establish in the island's capital, Hadibu. This arrival was not unexpected, since mainland Yemen, especially the city of Aden, has well-established populations of house crows originating from founders released by the British colonists at the end of the 19th century. The spread of house crows by ship across the region often reported (Kinnear 1942; Jennings 2004; Ryall 2008), but despite the negative effects of the crows in Aden and on the mainland (Ash 1984), there was no attempt at port sites to prevent the species arriving and establishing on Socotra. Furthermore, there was no rapid response to eradicate the newly-arrived birds on Socotra.

Table 1 Summation of dates and the population status of the house crow (*Corvus splendens*) on Socotra Island, Republic of Yemen.

Date	Status/action
1995/96	Pair of birds arrive on a ship
1998	Bounty payments started
2002/03	Population reaches 23 breeding birds
1998 - 2008	More than 550 chicks/eggs removed
2008	Bounty payments stopped
April 2009	13 birds killed. Population eradicated

Pages 361-363 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

The birds settled in a valley planted with palms along the edge of Hadibu. The valley contains a shallow stream arising from the interior mountains and running into the sea in the north. The character of the area is rural, commonly with gardening and domestic animals in the backyards of houses. The stream has considerable garbage pollution along its banks, making it an ideal environment for commensal species like house crows.

The birds nested in tall palms next to houses where all resources needed by the birds were available. Without any natural enemies, the crow population increased, leading SEPA in 1998 to instigate a bounty system as a means of restricting the rate of expansion. Increasing amounts of money was paid to teenagers for climbing to the nests and removing the chicks and eggs. Over ten years, more than 550 chicks and eggs were removed making this an effective method of control that kept the numbers very low. However, the method was costly and did not achieve eradication. At its peak, the house crow colony on Socotra comprised 23 breeding birds (Omar Al-Saghier pers. comm.).

In 2008, when bounty payments were stopped, the house crow on Socotra Island had potential to increase unrestricted. Studies elsewhere indicated that a population of 100 crows could reach 2000 within four years (Ali 2003). Concern about the growing impact of the house crow on native bird species (e.g., Ryall 1992a) then led to the decision that eradication of the population was necessary.

During this period, eradication attempts of trapping by SEPA personnel and shooting by marksmen from the Yemeni army had not resulted in any bird being killed. It was recognised that no abilities for eradication existed on the island or within the country. The use of foreign expertise was the next step, and a cooperative project aiming for the rapid, successful eradication of the house crow from Socotra was founded by SEPA at the end of 2008 supported by an financial contribution of the Small Grant Scheme of the Global Environment Facility (SGP/GEF), Yemen. The crow population at this time was estimated as 12 birds.

METHODS

This project was limited by financial constraints and visas were restricted to two weeks on the island by the foreign experts involved. All planning was through remote communication as neither of the two foreign individuals in the project team had been to Socotra so had no impression as to the exact situation. There are few comparable operations to eradicate extant populations of house crows and this limited previous experience to draw on when the operation was planned.

Poisoning with avicides like Starlicide (also called DRC1339, 3-chloro-ptoluidine hydrochloride) is the most commonly used technique to kill house crows in larger numbers on mainland Yemen (Jennings 1992). This method would have required more than the two weeks available and, due to the presence of large populations of two species of vultures, no poisoning was permitted.

Trapping would also have required more than two weeks. Also some of the birds were possibly trap-shy as a result of the failed trapping efforts by SEPA.

The option to shoot all of the birds was agreed to by all parties as the only available method. The shooting had to be by someone who was an experienced marksman and hunter, had worked on eradication projects for other species, and who knew how to apply techniques that would keep the house crows naïve about the aim of the project for as long as possible.

Three different firearms and appropriate ammunition were brought to Socotra. These were selected by the hunting expert based on years of experience of shooting crows in other parts of the world. The import of silenced .17 HMR and .22 R/F rifles, and a semi-automatic shotgun, were authorised by the Yemeni Interior Ministry.

The shooting was to be from a camouflaged window in an SEPA 4x4 Jeep. Senior SEPA staff were to be present at all times to guide the operation and talk to the public. The shooting team was also partially guided to locations by other observers. Occasional additional support from SEPA personnel was available.

RESULTS

The eradication project was conducted between mid to end of April 2009 on the outskirts of Hadibu. The local community supported the activities passively by not interrupting, and actively by showing where house crows had been seen, heard, or were feeding, roosting, and nesting. Residents became quickly aware of the fact that foreigners with guns were driving around in their neighbourhood. In recognition of the traditional, conservative, Muslim way of living in Hadibu, and the presence of weapons in most households, the permanent presence of SEPA (author of this paper) in the project team secured the safety of the shooter and provided explanations for reasons behind the activities to the local population.

Shooting began three days after the team arrived on Socotra. The first gun used was the silenced .22 calibre rifle with which half of the known population (six birds) was killed in one afternoon.

The crows then started to become more cryptic and careful. Although not yet able to identify the shooter, observer, or the car as a threat to avoid, the crows became less obvious. The next three birds were shot on day two, using a silenced .17 rifle and high power ammunition, which allowed shooting from the already necessary longer range.

After this, the three remaining crows were shy and partially started to leave the area for another valley 2-3 km away. The birds avoided staying at a site once the presence of the observer or the jeep was noticed. In order to discourage this wary behaviour after just two days of direct persecution by shooting, a day of observation was used to reduce stress on the crows. This also allowed time to re-count the remaining birds and identify possible shooting locations for the coming days.

On day four of the shooting operation the shotgun was used. The loud report made when firing this gun meant it was a less desirable tool in an urban or village setting. The first bird shot was intercepted flying between the two valleys. The second crow of the day was shot whilst a local person was climbing a known nesting tree to remove nesting material and/or eggs. From previous experience within the project, it was known that the crows would attack any human within the proximity of their nest and so the project team used this method to attract a bird to the site.

The last known bird (no. 12) was shot in the early afternoon, after two hours observation and identification of any patterns in its erratic flying and nervous behaviour. By then, the observer within the team had clearly been identified as a threat and the last crow kept its distance. As the bird was using the same palm fronds as look out posts, it allowed the shooter to get in position under one tree. The bird was then purposely driven by the observer toward the particular palm, using the "repellence-reaction" of the crow toward the observer. It was then shot.

After more than 500 man-hours of monitoring on foot, in cars, and from rooftops of houses, no further crows were seen, heard, or reported. An appeal was also put out within the local community for any crow sightings and an increased bounty was offered for any information. Seven days after the last known bird was shot, and just as the team was about to depart, a single crow was reported circling over the Hadibu Valley. SEPA personnel tried to find this bird's origin, as well as clarify its movement patterns. However they failed as the crow disappeared, returned two days later, then disappeared again. The specialist team therefore went back to shoot this last bird, which was seen as the most dangerous crow because its previous presence and origin were unknown as was the site to which it disappeared. There was a high likelihood that the bird was a single remaining nesting individual surviving in a neighbouring valley. After four days of observation and pursuit, the final bird (no. 13) was shot in Hadibu Valley, using magnum shotgun ammunition.

In total, after 15 days, 13 birds had been shot ending a 15 year old problem with the potential to become a major issue for the island's fauna and flora and people.

DISCUSSION

The initial action to control this invasive species was instigated soon after its arrival on Socotra. The use of bounty payments did slow the increase of the population but rapidly became too costly.

It is unclear why the crow population declined from the 23 bird peak in 2002/03 to 13 birds in 2009. The bounty system would have been slowing recruitment but the death of 10 adults is higher than expected. Birds of prey may have been having an unexpected impact.

The spread of the house crow is well documented and the bird is known for its abilities to populate new territories and survive under a variety of sometimes unfavourable conditions (Lever 2006). The success of the Socotra Island eradication can only be guaranteed when there is a system for rapid response to new incursions of crows. Otherwise, reinvasion will become an increasing risk as populations of crows expand in neighbouring countries or the wider region (Ryall and Meier 2008). Increasing ship traffic will likely add to this risk, although for the moment, due to piracy in the Gulf of Aden, this threat has temporarily decreased.

The best way of securing the results achieved on Socotra is to extend control or eradication into other areas. If control, or preferably eradication, of known house crow populations was strongly pursued elsewhere within the region, a system of sites without crow populations will develop. This would not only demonstrate that house crow control and eradication is possible, but more importantly provide immediate protection to native species and peoples' livelihoods. The reduction of populations in the region would also minimise the risk of birds reaching new areas or reinvading those already cleared. Well planned and coordinated approaches would address the spread of house crows through prevention, which is the most cost effective method of dealing with invasive species. However, at the remaining sites it will still need direct control to continue overall population reduction.

For example, to secure the achievements on Socotra, a small population of house crows in the port city of Salalah in Oman should be eradicated since many ships depart from this port to Socotra. The eradication of this population would secure a "crow-free" buffer zone for 1600 km along the Yemeni/Omani coast, minimising the chance of new populations building up there and enabling realistic monitoring for a "no-crow" zone.

Across the Gulf of Aden, in Somaliland, and the Autonomous Region of Puntland, there are newly detected, yet small populations of house crows. Their eradication would be comparably easy to implement since the populations are just a few dozen birds and security is much more advanced on the sites than in the neighbouring, former Somalia.

Such activities will buy the time needed to take on the larger populations of house crow in Djibouti and Eritrea on the African coast, but especially those in Aden and mainland Yemen on the south of the Arabian Peninsula. Significant funding and a work force need to be assigned for those tasks and of course there will also need to be a secure working environment. Eradicating house crows from Aden will not be an easy task since the birds are well established. Nonetheless, if the crows were eradicated from this area, major populations of crows in the region would finally be removed, and other small scale operations in the regions would achieve success without facing a permanent and increasing risk of reinvasion.

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Eradications of vertebrate pests from islands around New Zealand: what have we delivered and what have we learned?

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Abstract Eradications of invasive mammals have become increasingly complex and expensive. Increased public exposure and involvement in decisions about island eradications mean that conservation scientists must be prepared to justify the benefits of proposed eradications and defend the science used to measure cause and effect of agents of decline. Here I assess the biological, scientific and political outcomes of eradications on those islands in New Zealand from which all introduced mammal populations have been removed. By 2010, 147 populations of 13 species of vertebrates had been removed from a least 95 islands with a total area of 32,000 ha. Identified benefits to biodiversity were through *in situ* recovery, translocations or metapopulation management on the islands. These include improved prospects for 16 species of invertebrates, two species of frogs, three taxa of tuatara (*Sphenodon* spp.), 23 species of lizards, 32 taxa of terrestrial birds and 16 taxa of seabirds. The eradications can also be used to test hypotheses about the impacts of invasive species on native ecosystems. Considerable effort has been applied to understanding the effects of Pacific rats (*Rattus exulans*). There are now published accounts of the effects of these rats on plants, lizards, tuatara and seabirds, often using well designed field experiments. However, the effects of most other invasive vertebrates are poorly documented. Furthermore, impressive accounts of biodiversity achievements obscure potential problems. These include the genetic effects of small relict populations or small founders from translocations. Nonetheless, there has been acceptance of the value of these eradications at the highest political levels, government support for assistance in developing countries, and global export of technologies developed. A deeper understanding of the effects of invasive species, good reporting systems, and frequent communication and defence of benefits will be needed to gain public acceptance of increasingly ambitious projects.

Keywords: Biodiversity benefits, birds, reptiles, amphibians, invertebrates, plants, cause and effect, invasive mammals

INTRODUCTION

Invasive species are now recognised as major agents of global change (Mack *et al.* 2000; Simberloff 2003). The effects of invasive species are particularly severe on islands (Paulay 1994) where they are implicated in two thirds of recorded animal extinctions (Cole *et al.* 2005). On the other hand, there are increasing numbers of successful eradications, especially of introduced mammals. These include exotic foxes from 40 islands covering 210,000 ha in the Aleutian Islands of Alaska (Ebbert and Byrd 2002), 45 populations of introduced mammals from 29 islands in northwestern Mexico (Aguirre-Muñoz *et al.* 2011) and 21 species of introduced mammals from 17 islands in the Galapagos archipelago off Ecuador (Donlan *et al.* 2003). The upper limits of areas attempted have risen greatly since the 1990s (Donlan and Wilcox 2008). The eradication of mice (*Mus musculus*) and ship rats (*Rattus rattus*) is now being attempted on 3881 ha Rangitoto-Motutapu Island, New Zealand (Griffiths 2011), and the eradication of Pacific rats (*Rattus exulans*) has been achieved on 3083 ha Hauturu (Little Barrier) Island, New Zealand (Towns *et al.* 2006); Norway rats (*R. norvegicus*) on 11,300 ha Campbell Island, New Zealand (McClelland and Tyree 2002); cats (*Felis catus*) on 12,800 ha Macquarie Island, Australia; rabbits (*Oryctolagus cuniculus*) on 3450 ha Norfolk Island, Australia; and in the Galapagos Islands, Ecuador, goats (*Capra hircus*) on 458,812 ha Isabela Island, and pigs (*Sus scrofa*) on 58,465 ha Santiago Island (Donlan and Wilcox 2008).

Eradications on large islands are expensive and are likely to include sites with a high public profile, or inhabited by people. For example, since 1996 the Department of Conservation has undertaken ten large and complex island eradication campaigns at a total cost of over NZ $8 million (updated from Broome 2009). Among these, there was intense debate within the scientific community and Maori tribal groups (iwi) over the removal of Pacific rats from Hauturu in 2004 (e.g., Kapa 2003; Towns *et al.* 2006), which incurred legal costs of at least NZ$ 200,000 (Broome 2009). Elsewhere, eradication attempts have been stiffly resisted on the grounds of unacceptable collateral damage or concerns from animal rights activists (Towns *et al.* 2006). In the UK, a US$1.6 million attempt to remove hedgehogs (*Erinaceus europaeus*) introduced by the inhabitants of the Uist Islands of Scotland proved ineffectual – at least in the initial years – largely because animal rights activists convinced Scottish Natural Heritage to use live capture and relocation rather than kill trapping (Carrel 2007; Webb and Raffaelli 2008). Such examples pose a dilemma. Because of the extent to which invasive species can disrupt ecological processes and human welfare (Mack *et al.* 2000), increasingly ambitious eradications of these species should be attempted (Simberloff 2002). But as the public profile of these attempts increases, so does resistance to them, despite likely benefits to biodiversity, native ecosystems, and ultimately human welfare.

Since 1996, eradications requiring toxins in New Zealand have often been publicly notified through the Resource Management Act 1991 (RMA). Proposers must compile an Assessment of Environmental Effects (AEE), which is available for public submissions. The AEE and submissions are then examined by independent commissioners who may reject the application or place conditions on the way the project is conducted. Two key biological questions often arise during this process (pers. obs.). Firstly: "Do the benefits to biodiversity outweigh financial and short term environmental costs?" Secondly: "How good is the evidence for cause and effect between losses of biodiversity and purported agents of decline?"

Neither question is exclusive to eradication attempts on islands. Any attempted eradications should include measures of the benefits to species and ecosystems. In addition, treating the eradications as large-scale experiments should over time illustrate the relationship between introduced organisms and those that they affect (Towns *et al.* 1997).

In this review I describe the outcomes of eradications of vertebrates from islands around New Zealand and ask how measured outcomes have informed our understanding of the effects of invasive species. I first summarise the biological benefits attributable to eradications on islands from which all vertebrate pests have been permanently removed. I then

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Table 1 Number of invasive vertebrate populations removed from 95 islands around New Zealand, their general effects on native biota (King 2005) and the type and quality of evidence of their effects on island ecosystems.

Introduced species	No. Ops	Single pest	General diet in New Zealand	Evidence for effects	References
Weka *Gallirallus australis*	3	1	Invertebrates, reptiles, ground-dwelling birds including seabirds	Between island comparisons; stable isotopes (seabirds)	Harper 2007
Brushtail possum *Trichosurus vulpecula*	3	0	Foliage, flowers, fruit and bark of >90 spp of native plants; extensive canopy defoliation; predation of invertebrates (e.g., large snails), eggs, nestlings and adult birds (including seabirds)	Forest canopy recovery after eradication	Atkinson 1992
Rabbit *Oryctolagus cuniculus*	12	2	Grasses and shrubs	Recolonisation by broadleaved coastal shrubs after eradication	Towns *et al.* 1997
Pacific rat *Rattus exulans*	42	26	Foliage, flowers, fruit, seeds and seedlings of forest plants; wide range of invertebrates; lizards; eggs and chicks of some birds	Between island comparisons of plants, reptiles and seabirds; exclosure experiments with plants; post eradication recovery of invertebrates, plants, lizards, tuatara and seabirds	Whitaker 1978; Atkinson 1985; Towns 1991, 2002, 2009; Towns *et al.* 1997, 2007; Pierce 2002; Campbell 2009; Campbell and Atkinson 1999, 2002; Rayner *et al.* 2007
Norway rat *R. norvegicus*	26	11	Foliage, fruit, seeds and rhizomes of plants; wide range of invertebrates, lizards; eggs and chicks of some birds	Observed post invasion declines of tuatara; post-eradication responses of forest plants	Newman 1986; Allen *et al.* 1994; Campbell 2002
Ship rat *R. rattus*	6	5	Fruits of native plants; wide range of invertebrates, lizards; eggs, chicks and adults of some terrestrial and arboreal birds	Post invasion declines of invertebrates and forest birds and bats; stable isotopes (forest bids)	Atkinson and Bell 1973; Harper 2007; Towns 2009
House mouse *Mus musculus*	13	4	Seeds of native plants; wide range of invertebrates; some lizards and birds	Between island comparisons of invertebrates, post-eradication responses by invertebrates and lizards	Newman 1994; MacIntyre 2001; Roscoe and Murphy 2005
Stoat *Mustela erminea*	7	5	Invertebrates, lizards and birds; introduced rodents and rabbits	Post invasion declines of birds	King and Murphy 2005
Cat *Felis catus*	8	1	Invertebrates, lizards, birds (esp. seabirds); introduced rodents and rabbits	Post invasion declines of birds; post eradication recolonisation by land and sea birds	Fitzgerald and Veitch 1985; Fitzgerald *et al.* 1991; Girardet *et al.* 2001, Veitch *et al.* 2004; K. Baird (pers comm.)
Pig *Sus scrofa*	10	1	Fruits and foliage of plants, wide range of invertebrates; frogs and lizards; ground-nesting birds and their eggs; introduced rodents and rabbits	Exclosures; recovery of seabirds post eradication	Harper 1983; Coleman *et al.* 2001
Cattle *Bos taurus*	3	0	Wide range of herbs, grasses shrubs and trees	None recorded	
Goat *Capra hircus*	10	1	Fungi, ferns, grasses and broadleaved shrubs and trees	Post invasion destruction of vegetation; diet analysis; post eradication recovery of plant communities	Sykes 1969; Parkes 1984; Brook 2002; Bellingham *et al.* (2010b)
Sheep *Ovis aries*	4	0	Grasses and some shrubs	Post removal recovery of native herbs and grasses	Dilks and Wilson 1979; Meurk 1982; Meurk *et al.* 1994

ask whether the eradications provide less obvious benefits through scientific knowledge, communication, political support and international uptake.

STUDY SITES

Islands used here are those beyond the range of natural recolonisation by the eradicated vertebrates. Successful eradications are those with no recolonisation for two years or more after the original campaign. A few islands have occasional incursions of mammals through natural dispersal, but if these are consistently eliminated on arrival, the site is regarded as permanently clear and is included in the analysis. Guidance about motives for eradications was obtained from legal status of the land, statutory plans and interviews with project managers. Evidence of the effects of invasive species was regarded as available if accessible with search engines such as the Department of Conservation library catalogue, Google Scholar and BIOSIS.

Up to 2010, all invasive mammals and one species of bird had been removed from 95 islands; a total of 147 populations of 13 species of vertebrates within an area of 32,000 ha (updated from data In: Veitch and Bell 1990; Clout and Russell 2006). Eradications on an additional 20 islands (total 4700 ha) of eight species of vertebrates have yet to be confirmed. The most frequently eradicated species were Pacific and Norway rats (Fig. 1), but also included one species of out-of-range flightless predatory bird and one arboreal marsupial (Table 1). Most of the remaining species were farm animals that became feral, although domesticated livestock removed from islands retired as farms were not included in these totals. Assessments of the effects of feral species were complicated by the previous presence of stock on 20 (21%) of the islands, which in most cases were also cleared of forest for agriculture. Additionally, even the forested islands were burned during Maori or early European history (Bellingham *et al.* 2010a), although they have now had many decades to recover. Furthermore, on 25 (26%) islands, multiple species of terrestrial vertebrates coexisted, with potential for complex interactions between them (e.g., Courchamp *et al.* 1999, 2000). On the other hand, for most of the earlier eradications, multispecies removals were conducted over long time intervals, with the potential to measure responses between the eradications. Finally, all of the islands are inhabited by introduced birds such as European starlings (*Sturnus vulgaris*) and blackbirds (*Turdus merula*) whose effects are unknown. Many such species are now found through the entire archipelago and are assumed to have equal effects across the sample.

BIOLOGICAL OUTCOMES OF ERADICATIONS

Species and communities

Given that eradications were designed to protect and enhance depleted biodiversity, what were the benefits? Based on assessments of eradications over the last 20 years, *in situ* recovery or subsequent translocations to islands now free of introduced mammals around New Zealand improved the long term prospects for at least 16 species of invertebrates and 76 species of vertebrates. The latter included two of the four species of frogs, all three taxa of tuatara, 23 of the 80 species of lizards, 32 of the 73 taxa of terrestrial birds and 16 of the 84 taxa of seabirds (Bellingham *et al.* 2010a). Furthermore, earlier eradications of goats from Great Island (Three Kings Group) may have enabled the recovery of more than 200 species of plants and up to 30 species of endemic snails (Brook 2002; P.J. de Lange pers comm.; Bellingham *et al.* 2010b). Similarly, the removal of pigs from Aorangi Island (Poor Knights Group) likely provided benefits for numerous rare species, including 18 species of plants, five species of snails, 13 species of insects, six species of reptiles and two species of birds (Towns *et al.* 2009b; Bellingham *et al.* 2010a).

For many species, range contractions have been reversed after eradications as species are either returned to sites they previously occupied or released into new ones as a conservation measure. Excluding planting for island reforestation, translocations alone have involved at least 139 populations of 63 taxa of animals (Fig. 2). The results of species translocated to or between islands must be treated with caution because determining the success of translocations can be difficult. If we use self-sustaining populations as the minimum criterion for success (e.g., Dodd and Seigel 1991), birds have the highest proportion of identified successful translocations to islands after pest eradication 44/72 (61%). The proportion is much lower for invertebrates 3/21 (14%) and reptiles 3/37 (8%). None of the populations of amphibians and seabirds translocated to new islands can yet claim to have met basic criteria for success. In part, lack of data on success relates to the ease of locating released animals. With the exception of terrestrial birds, which often have flexible and high reproductive output, many invertebrates and reptiles are cryptic and difficult to locate at low density. Some, such as tuatara, also have low reproductive output and late age at maturity (Cree 1994). For such species the outcome of translocations may not be measurable for years or even decades after release (e.g., Towns and Ferreira 2001).

Furthermore, aside from at least three known failures (4%), there are also populations (all birds) that are maintained in island environments where they are unlikely

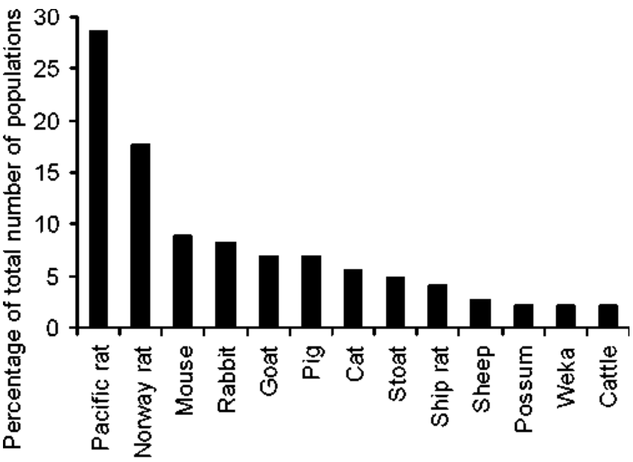

Fig. 1 Composition of 147 populations of invasive vertebrates removed from 95 islands around New Zealand.

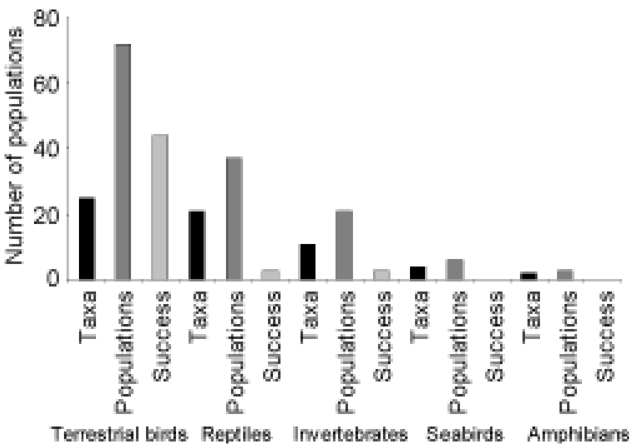

Fig. 2 Composition of 139 translocations of 63 taxa of native vertebrates and invertebrates to islands cleared of all introduced mammals.

to ever form self-sustaining populations, but where their prospects can be improved away from introduced predators. Examples of these include kakapo (*Strigops habroptilus*), kiwi (*Apteryx* spp.), takahe (*Porphyrio mantelli*) and hihi (*Notiomystis cincta*). Here success is based on overall increases in metapopulations, even though contributing populations may be very small (see also Bellingham *et al.* 2010a).

Populations that are expanding after invasive species removals may carry a legacy of past problems. For example, when Pacific rats threatened populations of northern tuatara (*Sphenodon punctatus*) on Hauturu, the remaining eight adults were taken into captivity to breed until Pacific rats were eradicated in 2004. Since 2006, over 100 tuatara raised in captivity have gradually been released (MacAvoy *et al.* 2007). This appears to be an exemplary breeding programme but the adult tuatara on Hauturu have lost genetic variation, with potential attendant problems of low fitness (MacAvoy *et al.* 2007; Miller *et al.* 2008). Furthermore, around 78% the released progeny were sired by one male (Moore *et al.* 2008). Tuatara can take over 10 years to reach sexual maturity and each female has an annual reproductive output of about 2 offspring (Cree 1994). Consequently, even determining numerical success or failure of the Hauturu population may take many decades. Establishing the genetic effects of a predation bottleneck and restricted paternity on tuatara may take even longer.

Similar problems can arise in translocated populations. Miller (2009) assessed the genetic heterozygosity of three populations of translocated lizards, each of which had self-sustaining populations (*sensu* Dodd and Seigel 1991). She found that when the founder population is low (15), or in larger populations when there is relatively low founder survival, inbreeding depression can erode genetic diversity sufficiently to jeopardise the long term prospects for the populations.

Such problems aside, natural recovery *in situ*, recolonisations, and translocations can greatly change the structure of communities on islands once invasive species have been removed. Some of these changes are subtle. For example, on Korapuki Island, lizard assemblages in the presence of Pacific rats and rabbits were dominated by diurnal species of skinks. After the two mammals were removed, dominance within the assemblages shifted as previously rare nocturnal geckos become increasingly abundant (Towns 1991, 2002). Similar subtle effects of rats such as Pacific rats have been reported for plant communities. Comparisons of seedling composition on islands where Pacific rats are present, have been excluded using cages, and have been eradicated, indicate that the rats have measurable effects on at least 11 and perhaps over 30 species of coastal and forest plants. These effects are sufficiently severe to result in impaired recruitment, sex imbalances and declines to local extinction of canopy and subcanopy species (Campbell and Atkinson 1999, 2002; Campbell 2011). There may also be a feedback loop, where predation on the large seeds of some plants by Pacific rats reduces their incidence in the canopy, thereby reducing visits from fruit pigeons (kereru: *Hemiphaga novaeseelandiae*) and dispersal of large-fruited plants that remain (Campbell and Atkinson 2002). The extent to which changed seedling recruitment after release from the effects of Pacific rats might change forest composition is as yet unclear.

More extensive changes in community structure can follow the removal of grazing species such as sheep and goats. On subantarctic Campbell Island, removal of sheep from the island in 1990 was followed within four years by recovery of tall native grasslands, reinvasion of the old pasture by native megaherbs, and declines in coverage by native species resistant to grazing. Full recovery of native plant communities is likely within a few decades (Meurk *et al.* 1994). Likewise, after the removal of goats from Great King Island in 1946, grazing-induced turf was 40 years later replaced by early successional forest up to 2m tall, and reappearance in coastal forest of endemic tree species (Wright and Cameron 1990; Bellingham *et al.* 2010b). However, the spread of some endemic species has been slower than expected, largely due to the absence of birds able to disperse large seeds. The importance of dispersers was illustrated when the translocation of a small number of kereru to Great King Island was rapidly followed by the appearance of new populations of seedlings (Bellingham *et al.* 2010b).

Ecosystems and landscapes

The removal of invasive species should, in theory, enable the recovery of ecosystems dominated by native species (Towns *et al.* 2009b). In New Zealand, 47(49%) of the eradications were on island Nature Reserves, where the removal of exotic organisms is mandated in order to protect the integrity of native ecosystems (New Zealand Reserves Act, 1977). However, measuring ecosystem responses to eradications has proved challenging. Recent advances centred on the role of seabirds as drivers of island ecosystems (Towns and Atkinson 2004; Bellingham *et al.* 2010a). On islands off northeastern New Zealand, Fukami *et al.* (2006) compared ecosystem processes on islands with large seabird populations with those where seabirds are suppressed by rats. The authors found that compared with islands invaded by rats, soils on seabird islands had higher total C, total N, total P, and marine-derived $\delta^{15}N$, greater microbial CO_2 production, and more abundant herbivorous and microbe-feeding nematodes. Many macro-invertebrates in the forest litter were also more abundant on seabird islands, including such diverse groups as beetles, collembolans and minute land snails (Fukami *et al.* 2006; Towns *et al.* 2009a). The seabird effects were also reflected in higher foliar and litter N concentrations, greater N to lignin ratios and higher litter decomposition rates (Wardle *et al.* 2009). In contrast, compared with the islands invaded by rats, seabird islands had lower seedling densities and lower tree basal area, reflecting the disturbance effects of seabird on forest vegetation (Fukami *et al.* 2006; Roberts *et al.* 2007). These observations were tested experimentally by Jones (in press), who added fertiliser to mimic guano on Maud Island, which has few seabirds. As previous comparative studies indicated (e.g., Fukami *et al.* 2006), the treated sites had increased litter decomposition rates, arthropod consumer abundance, and above-ground net primary productivity. Jones (2010a) also measured $\delta^{13}N$ and C:N ratios in soils, plants and spiders on northeastern islands with expanding seabird populations 12–22 years after Pacific rats had been eradicated. She found that the two measures of N increased with time, indicating that these islands would converge with equivalent measures on uninvaded islands within about four decades.

On some islands, the removal of invasive species of large herbivores has led to changes of entire landscapes. On Campbell Island, Meurk (1982) described the rapid reappearance of brightly-flowered megaherbs in areas protected from sheep. Previous examples documented succession from turf to forest on Great Island after the removal of goats. Similar landscape-level changes are now apparent on islands retired from grazing and planted by volunteers. For example, when farming ceased on Tiritiri Matangi Island in 1971, only 11% of the original forest cover remained. By 1994, 280,000 trees had been planted (Rimmer 2004), and at least 60% of the island now has a rapidly closing canopy of young forest (R. Renwick pers comm.).

SCIENTIFIC AND POLITICAL OUTCOMES OF ERADICATIONS

Science and communication

When eradications are proposed, conservation scientists are frequently asked: to provide evidence that species proposed for removal have detrimental effects on native species. Post-eradication studies of recovery by native species and communities should provide strong evidence of the effects of introduced species (Veltman 1996; Towns *et al.* 1997; Towns 2009), particularly if only one pest species was present. To examine where understanding has advanced, I have listed all 13 species eradicated from the 95 islands, identified the general effects on New Zealand biota based on recent reviews, and then identified how island studies have contributed to this information (Table 1).

The most comprehensive studies have been on the effects of Pacific rats, where distributional comparisons of plants were the basis for hypotheses tested by exclosures and post-eradication responses for plants (discussed above). Among vertebrates, hypothesised direct effects of Pacific rats on the eggs and hatchlings of tuatara were confirmed when there was a pulse of tuatara recruitment after the removal of Pacific rats from three islands (Towns *et al.* 2007). However, there was also an unexpected indirect effect, where some populations of adult tuatara also showed significantly better body condition (length: mass) when Pacific rats were removed – presumably due to release from interference competition (Towns *et al.* 2007). These more subtle effects also became apparent for resident burrowing seabirds with rapid increases of fledging success when Pacific rats were removed (Pierce 2002; Imber *et al.* 2003; Rayner *et al.* 2007; Towns 2009).

Aside from useful studies of the effects of invasions by cats and sheep (Table 1), there are few detailed accounts of responses after removal of some of the most widespread pest species. For example, a lack of comprehensive post eradication monitoring after the removal of pigs represents a missed opportunity to inform debate about their effects on native species other than seabirds. For species such as ship rats, the short history since eradication may account for the lack of published information on responses by native species. On the other hand, there has been only one study of the responses after removal of Norway rats, despite a long time interval and numerous potential study sites.

The sparse examples of benefits of eradications supported by peer reviewed articles in reputable scientific journals is one reason for conflict between conservation organisations advocating pest eradication and sector groups in opposition (Towns *et al.* 2011). In one example, animal rights activists attempted to use court action followed by direct sabotage in an attempt to terminate the eradication of ship rats from Anacapa Island in California. The activists argued that conservation benefits did not outweigh the collateral costs to native species, the rats had been demonised and the eradication was being undertaken only because the rats were there (Towns *et al.* 2006, H. Jones pers comm.). Correspondents in New Zealand can hold similar views. One recent letter to a newspaper complained of this "demonising" attitude inherent in the Department of Conservation's attempts to remove hedgehogs (among six other species) from Rangitoto and Motutapu Islands. Public attitudes to invasive species in Scotland were shaped by awareness and education (Bremner and Park 2007), which suggests that some opposition to eradications stems from poorly developed proposals. In New Zealand, other than in rare examples where eradications were undertaken on small islands to test methodologies, they were all done with a view to protect threatened species, enable the public to experience prolific native wildlife on islands without introduced pests and to restore modified ecosystems (Broome 2009). Such aims can be difficult to communicate if the media prefers stories about the conflict or complexities generated by projects instead of their benefits (Bremner and Park 2007).

Surprises and failures

A question I have sometimes been asked at RMA hearings is: "Are there detrimental long term effects of eradications on island species?" There are few such examples. Perhaps the most notorious is the invasion of native plant communities by invasive boxthorn (*Lycium ferocissimum*) after the removal of rabbits from Motunau Island (3.5 ha) in 1963. The thickets of this spiny shrub became so dense they were responsible for entanglement problems for nesting seabirds (Beach *et al.* 1997). More often, the unpleasant surprises have been less vigorous response by native species than expected. One example is the slow spread of species with large seeds after removal of goats from Great Island (see above; Bellingham *et al.* 2010b). Another was a lack of measurable response by forest birds after the removal of cats from Hauturu Island (Girardet *et al.* 2001). This eradication did have the desired effect of reducing predation of adult Cook's petrels (*Pterodoma cookii*) by cats, but the unpredicted effect of increased predation by Pacific rats on Cook's petrel chicks until the rats were eradicated in 2004. The pressure on petrel chicks was attributed to mesopredator release, after removal of cats as a major predator of the rats (Rayner *et al.* 2007). The only other negative outcomes have eventuated from reinvasions of rats to islands. For example, three species of rats were eradicated from Pearl Island (512 ha) 225 m off Stewart Island. Although reinvasion by ship and Norway rats from Stewart Island was predicted, pre-eradication analyses of microsatellite DNA in both island populations indicated rare mixing between them. However, both species reinvaded after only nine months, with their origins on Stewart Island verified by microsatellite DNA (Russell *et al.* in press). Aside from the value of DNA analyses, the study demonstrated that the hypothesis of infrequent reinvasions by rats did not hold after the Pearl Island rat populations were eradicated.

The Pearl Island experience did provide a useful test of rat dispersal capabilities, even though the outcome was disappointing. For most other eradications, surprises have been more positive, including rapid and unpredicted recolonisations by native species. For example, three species of native birds recolonised Rangitoto-Motutapu Islands within 12 months of aerial bait spread against vertebrate pests and before the full programme had been completed (R. Griffiths pers. comm.).

National and international support for island eradications

The development of increasingly effective methods against invasive mammals such as rodents and cats (Veitch 2001; Thomas and Taylor 2002; Towns and Broome 2003) has received political support at the highest levels in New Zealand. For example, the planning and execution of the campaign against Norway rats on Campbell Island was cited for Innovative Practice in the New Zealand Public Service. The proposal to remove all seven species of invasive mammals from Rangitoto-Motutapu Islands was announced by the then Prime Minister and Minister of Conservation (Clark 2006). Internationally, even the earlier successes were seen as so important that a squad of New Zealand eradication experts was proposed to assist other nations with removing threats to their biodiversity (Duffy 1994). New Zealand has become an acknowledged leader in island conservation (e.g., Rauzon 2007), and pest eradication was even identified as one of New Zealand's "export industries" (Simberloff 2002). For example, New Zealand advice, assistance and specialised equipment been used in such diverse locations as the Seychelles, Falkland Islands and Western Australia (McClelland and Tyree 2002). New Zealanders have also assisted with the eradication of rabbits and ship rats on the French island of Saint Paul (Micol and Jouventin 2002), ship rats on San Pedro Mártir, Farallón de San Ignacio and Isabel Islands in the Gulf of California, Mexico (Samaniego-Herrera *et al.* 2009), Norway rats on Rat Island in the Alaskan Aleutians (S. Buckelew pers comm.), pigs on Santa Cruz Island off California (Parkes *et al.* in press) and goats on Lord Howe Island off Australia (Parkes *et al.* 2002). Furthermore, the New Zealand government through NZAID supports the Pacific Invasives Initiative, a non-government organisation that facilitates capacity development and provides project management advice for the eradication and control of invasive species throughout the Pacific region (http://www.issg.org/CII/tools.html).

CONCLUSION

Eradications of invasive species are no longer novel; they are increasingly ambitious and expensive, which also makes them increasingly difficult to justify unless there are unequivocal benefits (Simberloff 2002). The New Zealand public has gained increasing involvement in the choice of sites for eradications, the restoration of island systems cleared of pests, and in the eradication methods used (Towns *et al.* 2011). This involvement increases the need to answer questions about the outcomes of eradications, cost-effectiveness, and the effects of invasive species on native species and ecosystems.

The outcomes of eradications can be measured in two ways. The first involves tangible measures: the rate of recovery of resident species, recolonisation by extirpated species, reappearance of species reduced to undetectable levels, and the effectiveness of reintroductions of species unable to disperse to newly available sites. The measures can become increasingly complex as responses affect communities, ecosystems, and landscapes. There are also intangible measures: the effectiveness of communicating results to the scientific community and the public, political acceptance of the methods used and benefits gained, and the export of technologies to other locations. The two groups of outcomes are linked. Without the intangible measures such as political support, the management of invasive species cannot proceed. Although examples of all such outcomes are provided here, data for some of the most straightforward measures were difficult to obtain. Even with a long history of eradications in New Zealand, records of biodiversity gains are often buried in grey literature and reports to local conservation offices. For example, of 86 reports of reptile translocations around New Zealand, only 15 (17%) were in the primary literature (Sherley *et al.* 2010). Given that the available data under-estimate achievements, the scientific community can only communicate to the public rather vague views of the extent of change possible. We are also a long way from measuring, or even identifying, the ecosystem services that ecological restoration can provide. As a first step, the more tangible measures could be assisted by regularly updated databases of successful eradications (e.g., Keitt *et al.* 2011) and, within defined criteria for success, a list of the species known to benefit.

In New Zealand, eradications of introduced vertebrates from islands were to some extent viewed as experiments since they effectively used a "learning by doing approach" that tested the technologies of removal (Thomas and Taylor 2002; Towns and Broome 2003, Broome 2009). Unfortunately, a similar developmental approach was not taken to measuring the outcomes of eradications. Had questions about the effects of specific introduced species been identified and costs of pursuing them included in the project from the outset, we would now be in a much stronger position to identify cause and effect. For the more recent eradications, especially of some species of rodents, retrospective analyses of the responses of resident species and ecosystems might still prove enlightening. But for others, such as the historic removal of pigs, any but the coarsest of analyses are now obscured by interactive effects and time.

Fortunately, despite the few published reports of gains from eradications, funding has so far been found for progressively more ambitious projects (e.g., Broome 2009). One test of the political will is whether such projects continue in the face of any public disquiet. In a recent example, the spread of baits against rodents was able to continue despite mistaken claims that deaths of dogs and marine life on the Hauraki Gulf beaches were an effect of eradication campaigns against the seven species of pests on nearby Rangitoto-Motutapu Islands (Morton 2009; Griffiths 2011).

Perhaps we should now invite our international colleagues to fill the gaps that we have left by providing more comprehensive and scientifically robust accounts of their efforts (Table 2). For example, although the benefits of pest eradication in New Zealand may seem impressive, they are now being matched elsewhere such as in Mexico (Aguirre-Muñoz *et al.* 2011). Where New Zealand may still contribute is from a temporal perspective, with its numerous locations where invasive vertebrates have been

Table 2 Summary of information needed on effects of the more abundant invasive species of mammals on islands around New Zealand.

Species	Existing knowledge	Information needed
Rabbit	Sparse available information confounded by other introduced species	Effects on island plant communities
Pacific rat	Extensive information on effects on plants, some invertebrates, reptiles and some seabirds; all data from northern islands	Equivalent studies for southern islands
Norway rat	Post-eradication responses of plants on one southern island (Breaksea) and some northern islands	Effects on invertebrates, reptiles and birds over wide geographic range
Ship rat	One invasion confounded by presence of weka (Big South Cape/Taukihepa); sparse post eradication data (Matiu/Somes)	Effects of plants and most groups of animals over wide geographic range
House mouse	Patchy data from one island (Mana)	Effects on vegetation, invertebrates and small reptiles
Cat	Some studies on forest and sea birds	Direct and indirect effects on island ecosystems
Pig	Anecdotal accounts except for seabird recovery on one island (Aorangi)	Direct and indirect effects on island ecosystems
Goat	One comprehensive study (Great King)	Indirect effects on island ecosystems

successfully removed for long periods (Jones 2010b). A focus on the natural, social and economic benefits of restoration of these island ecosystems could then become a particularly fruitful basis for international collaboration (e.g., Mulder *et al.* 2011).

In sum, New Zealand has a strong history of development of eradication technology, high levels of national political support and international influence, but patchy contributions to understanding the relationships between native species and agents of decline. This understanding would be improved if outcome monitoring, together with the collection of appropriate baseline data, were at the outset incorporated into project design and costs.

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Changes in bird numbers on Raoul Island, Kermadec Islands, New Zealand, following the eradication of goats, rats, and cats

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Abstract Raoul Island (2938 ha; 29°16'S, 177°52'W) is the largest island in the Kermadec Group and is situated 995 km from the nearest part of mainland New Zealand. It is the summit of a large and active volcano rising from the Kermadec Ridge. The forest on Raoul is dominated by Kermadec pohutukawa (*Metrosideros kermadecensis*) with an understory of broad-leaved fruit-bearing plants, ferns and palms. Prior to the introduction of browsing and predatory mammals, Raoul had an abundant seabird population and a limited landbird population of endemic and native species. Several exotic landbird species established following their introduction to mainland New Zealand in the late 1800s, which was also after the introduction of several species of mammals to Raoul. The introduced mammals reduced seabird populations to possibly only two species continuing to breed in low numbers on Raoul. The forest became a canopy with little understory. Some forest bird species declined in number while others increased; at least three species became extinct on Raoul. Following eradication of all the introduced mammals, seabirds are returning to the island; we report sightings of 11 seabird species on Raoul, with evidence of breeding in black-winged petrels (*Pterodroma nigripennis*), wedge-tailed shearwaters (*Puffinus pacificus*), Kermadec petrels (*Pterodroma neglecta*), white terns (*Gygis alba*), and sooty terns (*Onychoprion fuscatus*). Grey noddies (*Procelsterna cerulea albivitta*) and red-tailed tropicbirds (*Phaethon rubricauda*) are now roosting, possibly breeding, on Raoul. Great frigatebirds (*Fregata minor*) have been observed in numbers that suggest future breeding. Kermadec little shearwaters (*Puffinus assimilis kermadecensis*), Kermadec storm petrels (*Pelagodroma albiclunis*), and white-naped petrels (*Pterodroma cervicalis*) are prospecting. Some forest bird species have declined in number while others have benefited from improved forest condition.

Keywords: Monitoring, *Capra hircus*, Norway rat, *Rattus norvegicus*, Pacific rat, *Rattus exulans*, *Felis catus*

INTRODUCTION

Raoul Island (2938 ha; 29°16'S, 177°52'W) is the largest island in the Kermadec Group and is situated 995 km from the nearest part of mainland New Zealand (East Cape) (Fig 1). The island is roughly triangular in shape, approximately 10 km long and 7 km wide and rises to 516 m at Mt Moumoukai Its topography consists of a steep-sided central caldera with major ridges to the west and south from which run sharply dissected ridges and ravines.

A boulder and rock coastline flanked by cliffs up to 250 m in height surrounds most of the island, although sand and gravel beaches occur at Denham Bay and to a lesser extent on the north coast in front of Low Flat and the Terraces. Flat to undulating land is essentially restricted to Denham Bay, Low Flat, the Terraces and to the floor of the caldera. Three lakes occur on the floor of the caldera; the largest being Blue Lake, followed by Green Lake, and Tui Lake. The lakes are periodically affected by volcanic activity and do not provide a consistently potable water source. Standing water also occurs in the centre of the Denham Bay flat, and freshwater springs occur at the western end of the Terraces and on the coast north of Lava Point.

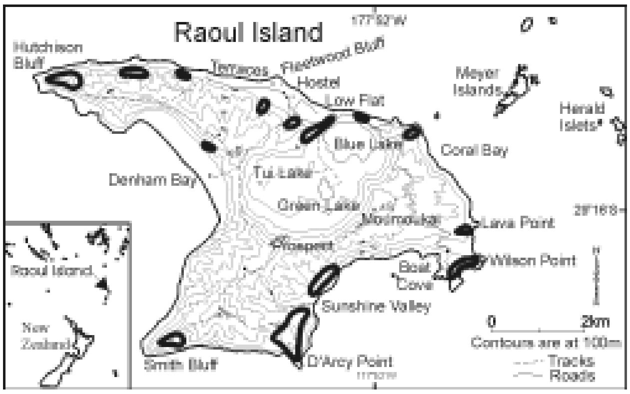

Fig. 1 Raoul Island. The areas marked in bold show the distribution of black-winged petrels as at February 2010.

Forest on Raoul Island is dominated by Kermadec pohutukawa (*Metrosideros kermadecensis*) with *Myrsine kermadecensis* and *Ascarina lucida* var. *lanceolata* as the predominant understory. Other common species include mahoe (*Melicytus ramiflorus*), wharangi (*Melicope ternata*), kawakawa (*Macropiper excelsum* var. *majus*), karaka (*Corynocarpus laevigata*), the tree ferns *Cyathea kermadecensis* and *C. milnei*, and nikau palm (*Rhopalostylis cheesemanii*) (Sykes 1977).

Raoul Island has a mild subtropical climate, with a mean annual temperature of 19°C and only small seasonal and daily temperature ranges. The maximum temperature recorded is 28.3°C, and the minimum 7.4°C; frosts are unknown. Rainfall averages 1535 mm, evenly distributed throughout the year (Anon 1979). South-easterly and easterly winds predominate in summer, and north-westerlies at other seasons (Williams and Rudge 1969).

Adjacent to Raoul are eight islands large enough to sustain vegetation, and a number of smaller stacks. These lie off the north-east coast of Raoul, and in Boat Cove (Fig 1). These islands are all free of introduced predatory or browsing mammals and there are no signs that they have been subjected to fire. Introduced weeds are present on the Meyer Islands.

Macauley Island (306 ha; 30°15'S, 178°32'W) lies 108 km south-south-west of Raoul. Curtis and Cheeseman Is. and L'Esperance Rock are further to the south.

The data presented in this paper come from occasional field expeditions and New Zealand Department of Conservation records. For the most part they were not gathered to specifically record the pre- and post-eradication bird populations, but rather as an ongoing record of the avifauna. Since 2007 a more determined effort has been made to document avian recolonisation and breeding populations through island-wide searches (Ortiz-Catedral *et al* 2009; Ismar *et al* 2010; Gaskin *in press*), and in conjunction with the New Zealand Department of

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Conservation (DOC) weed programme, and Raoul staff observations. The purpose of this paper is to provide as accurate as possible baseline data for future investigations, with respect to both seabirds and terrestrial species in the context of changes to an unusual bird fauna as it recovers from major biological disturbance through predation and habitat modification.

The Kermadec Islands have no indigenous land mammals or herpetofauna. On Raoul Island, Polynesian voyagers introduced Pacific rats (*Rattus exulans*), probably from the southern Cook Islands, possibly earlier than A.D. 1250, and, evidence suggests also at a later date, from New Zealand (Matisoo-Smith *et. al* 1998, 1999). Cats (*Felis catus*) were established on Raoul by 1836 (Straubel 1954), and Norway rats (*R. norvegicus*) probably arrived when the schooner *Columbia River* was wrecked in 1921 (Ingram 1972; Merton 1968).

In the period between their introduction prior to 1836 (Straubel 1954) and their removal in 1972-85 (Parkes 1990), goats (*Capra hircus*) modified the vegetation considerably (Sykes 1969). They removed almost all natural understory, allowing little or no regeneration of canopy species, and permitting dense stands of the introduced aroid *Alocasia brisbanensis* to flourish. Many coastal slopes became grasslands. The significant reduction of goat numbers from the early 1970s allowed extensive regeneration of vegetation to occur. The bare parts of the forest floor became covered in a dense layer of litter (Fig 2). However, the continuing presence of rats and lack of seed-dispersing birds inhibited seedling growth and species diversity in most places (West 2011).

Domestic pigs and dogs have been present, but did not establish as feral populations. Both would have had an impact on ground-nesting birds.

Until the mid 1980s, the Terraces were grazed by sheep and cattle, but these have now been removed from the island. For the most part, the old farm is now rank grass, which provides little food or habitat for most birds. A small mown airstrip is utilised by a number of bird species.

Rats and cats were eradicated from Raoul Island in 2002 and 2004, respectively (Broome 2009).

The native forest avifauna of Raoul has strong connections to the avifauna of New Zealand. Since European colonisation of New Zealand, further forest and waterbird species have reached Raoul Island. The introduced mammalian predators, with forest modification by the goats, has changed the relative abundance of these species and caused the extinction of at least three species from Raoul Island (Veitch *et al.* 2004).

Nesting seabirds were extremely abundant on Raoul in the past. For example, Iredale (1910) recorded "immense numbers" of wedge-tailed shearwaters (*Puffinus pacificus*) and "about half a million" Kermadec petrels (*Pterodroma neglecta*). White-naped petrels (*Pterodroma cervicalis*) were also present at the time of Iredale's visit, but evidence of cat predation was notable (Bell 1910). By that time seabird populations are likely to have been greatly reduced by cat and rat predation, with smaller species either extirpated or severely reduced (Gaskin *in press*). Seabird chicks and eggs were also harvested for food by settlers and visiting sailors up to the 1930s (Bacon 1957); even their down and feathers were used to stuff pillows and mattresses (Large 1888).

By the end of the twentieth century Raoul was practically devoid of seabirds (Veitch *et al.* 2004).

By 2000, Kermadec petrels, white-naped petrels, and Kermadec storm petrels were not recorded on Raoul. Burrows attributed to wedge-tailed shearwaters, Kermadec little shearwaters (*Puffinus assimilis kermadecensis*), and black-winged petrels (*Pterodroma nigripennis*) were occasionally found but those that were checked for breeding activity were found to be empty. It is possible that a few red-tailed tropicbirds (*Phaethon rubricauda*) nested successfully on remote cliff-ledges; sooty terns (*Onychoprion fuscatus*) remained in small colonies on the northern beaches in the 1990s; and a few white terns (*Gygis alba*) could be seen along southern coasts and at the forest edge behind the northern terraces.

METHODS

Forest Bird Counts

Forest bird counts on Raoul Island were instigated by Don Merton in January 1967 (Merton and Veitch 1986) during the Ornithological Society of New Zealand visit and have been repeated by Dick Veitch in 1994, 1998 (Veitch 2003), and 2008. The counting protocol consisted of one minute stops and four minute walking counts along each transect. All birds seen or heard within 100 metres were counted. Each transect was counted once in each year.

In 1967 these transects were three hour walks on two routes south of Mt. Prospect (Fig. 1) and along the Boat Cove Road. Forest changes following goat eradication made a repeat of the Mt. Prospect transect impossible in 1994, so a track that was cut between Trig V and the Hutchinson Bluff Track (the Top Track), and the Boat Cove Road were counted instead. These two transects were also used for the 1998 and 2008 counts.

The time of year when the counts were made has varied: January 1967; June/July 1994; July 1998; March/April 2008.

Seabird Observations

Since eradication of all mammalian predators and pests by 2004, surveying for seabird breeding has been undertaken spasmodically, with evidence gathered by Department of Conservation (DOC) staff during weeding programmes and casual hiking expeditions. There has also been an annual sooty tern survey (Potier and Shanley, Internal DOC report, 2009), and by K. Baird (KB), S. Ismar (SI), and C. Gaskin (CG). Surveys of known black-winged petrel and wedge-tailed shearwater colonies and more general island-searches to find breeding seabirds were undertaken during visits from October 2006 to April 2008.

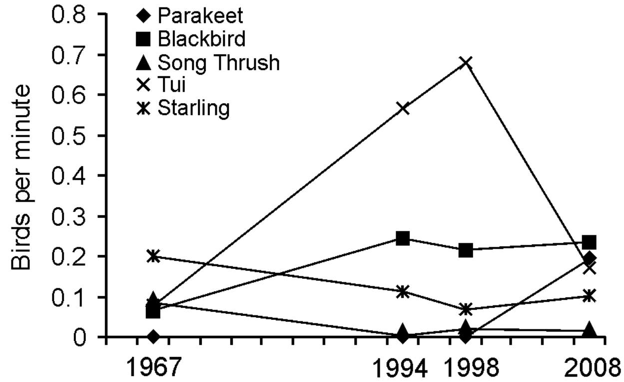

Fig. 2 Forest birds counted (mean number per minute)on Raoul Island. Note that counts were undertaken at different times of the year: Jan 1967, Jun/Jul 1994, Jul 1998, and Mar/Apr 2008.

RESULTS

The data obtained from the four sets of forest bird counts from 1967 to 2008 are shown in Fig. 2. No statistical analysis of these counts was possible. Various methods were tested to portray the data and all resulted in showing the general picture. This is a good portrayal of the counts but not a realistic record of forest bird numbers, particularly in 2008. Details of each species are included in the species accounts below.

Wedge-tailed shearwater (*Puffinus pacificus*). The first evidence of wedge-tailed shearwater breeding on Raoul after predator eradication was in May 2007, when two live chicks close to fledging were found on the beach near Fleetwood Bluff. Eight burrows, at least five of which had been active, were subsequently detected in the cliffs at this area. By 2008, the number of burrows at this site had increased to eleven, with eight active as judged by guano splashes around the entrances. Burrow entrances could be seen on cliffs west of the initially detected colony, but it could not be confirmed if they held chicks or had been frequented by adults. An additional breeding site was found a little further to the west, with seven burrows, at least five of which were holding chicks (SI, CG). In April 2008, a wedge-tailed shearwater was found prospecting at the entrance of one of the black-winged petrel burrows on the Hostel Cliffs (SI).

Kermadec little shearwater (*Puffinus assimilis kermadecensis*). Seen flying near the Hostel and one individual was found in the guttering of the Hostel in September 2007 (DOC staff notes).

Black-winged petrel (*Pterodroma nigripennis*). DOC volunteers first detected re-colonising black-winged petrels in the Coral Bay and Crater Lake Track areas on Raoul Island in 2006. After that, four breeding areas of the species were located in a survey in May 2007. By this time, the black-winged petrel had established breeding colonies on the slopes over Coral Bay, in woody areas around the Crater Lake Track, on the grassy northern cliff faces close to the Hostel, and in grassy areas behind the Hostel (Ismar *et al*. 2010). By March 2008, these breeding areas had extended, except for Coral Bay. New burrows were found at many locations along the northern slopes (Ismar *et al*. 2010). DOC staff camping at D'Arcy Point in 2008 reported many black-winged petrel burrows and birds landing amongst them at night. In 2008, CG found burrows at Smith's Bluff, Wilson's Point and Hutchison's Bluff. Birds were seen entering forest on ridges in the vicinity of Sunshine Valley and D'Arcy Point. In January 2010, DOC staff reported finding new burrows across the northern slopes during their weeding programme (SI). The known distribution by January 2010 is shown in Fig. 1.

Kermadec petrel (*Pterodroma neglecta*). Large numbers of summer-breeding Kermadec petrels formerly bred on Raoul Island. There has been one recent record of breeding with a large chick found at Nash Point on Raoul Island in September 2006 (DOC staff notes).

White-naped petrel (*Pterodroma cervicalis*). Now confined as a breeding species to Macauley Island but individuals of this species were recorded in February 2005 and 2006 caught in velcro grass (*Cenchrus calyculatus* Cav.) on the northern terraces of Raoul Island (DOC staff notes). It is also known from at-sea observations to be in waters around Raoul Island in May (Gaskin in press.).

Kermadec storm petrel (*Pelagodroma albiclunis*). Individuals flew onto the Hostel veranda on two separate occasions (29 May 2008 and 24 August 2008), indicating that prospecting is possibly occurring (DOC staff notes) or the birds were attracted to the Hostel lights.

Red-tailed tropicbird (*Phaethon rubricauda*). Thirty individuals were counted along the northern beaches and seen performing aerial display flights in 2007 (KB). In 2008 a similar number of birds, including some pairs, could be seen on cliff ledges leading to Hutchison Bluff, also performing aerial displays (CG, SI).

Great frigatebird (*Fregata minor*). This species has been reported in numbers (≤ 18 birds) (DOC Raoul staff Thirdly Reports); KB, CG (pers. obs. 2006, 2008) that suggest possible future breeding (G. Taylor, DOC, 9 June 2008 pers. comm.).

Spotless crake (*Porzana tabuensis*). This species was absent from Raoul in 1967 but is now present. In 2008, they were seen or heard in the dense grasses around the Hostel and along the back of the northern terraces. Spotless crakes have also been reported from the dense ferns behind the Denham Bay dunes.

Pukeko (*Porphyrio melanotus*) are now a common bird of the forest edges. In previous bird records they have either been present in low numbers or confirmed as absent (Veitch *et al.* 2004). They have increased in number since the removal of introduced mammals.

Sooty tern (*Onychoprion fuscatus*). Remnant populations probably remained on Raoul Island until cats were eliminated prior to 2004 (Broome 2009). In 1966/67 the Ornithological Society of New Zealand expedition estimated 40,000 pairs in Denham Bay and another 40,000 along the southern side of Hutchison Bluff (Veitch *et al.* 2004). By 1995 just 2230 birds were counted during the breeding season at Denham Bay, but by 1997 none were at Denham Bay and few were elsewhere on the coast of Raoul (Veitch *et al.* 2004). By 2006 sooty terns were breeding on the beaches to the north of Hutchison's Bluff and apparently expanding their colony each year. Estimates of population size by DOC were hampered by methodological problems and the desire not to negatively impact breeding birds, made difficult by the long narrow stretches of beach. An estimate was made during the 2008/2009 breeding season by two volunteers (Potier and Shanley, Internal DOC report 2009). Using a density estimate from quadrats where nests were counted and extrapolating for the measured size of the colony they estimated between 7634 and 9330 birds breeding on Raoul Island. There is no evidence yet (2010) of sooty terns returning to their former stronghold at Denham Bay (KB).

Grey noddy (*Procelsterna cerulea*). Possibly breeding, certainly use Raoul Island cliffs at Hutchison Bluff and Boat Cove for roosting (KB, CG, SI).

White tern (*Gygis alba*). Up to 12 individuals seen between Boat Cove and Sunshine Valley in 2007 (CG, KB) and chick-feeding observed on one occasion at Boat Cove (KB).

New Zealand pigeon (*Hemiphaga novaeseelandiae*), which were recorded by early settlers (Veitch *et al.* 2004) continue to be absent from the Kermadec Islands. There is now an abundance of food suitable for pigeons and they should now be considered for re-introduction to Raoul, as originally suggested more than 20 years ago (M. Clout pers. comm.).

Kermadec parakeet (*Cyanoramphus novaezelandiae cyanurus*). After the eradication of goats, parakeets were heard daily in the forest but numbers were low and breeding considered unlikely. There was a significant increase after the eradication of rats and cats, and breeding was proven in 2008 (Ortiz-Catedral *et al.* 2009). In 2008 they were present for the first time in the forest bird counts, but were quiet and very tolerant of counter presence. On many occasions they were seen to fly from the ground to perches three to five metres from the track and just sit there quietly while the counter passed by.

Long-tailed cuckoo (*Eudynamys taitensis*) were seen during the April 2008 visit, one bird was seen in clear view at Denham Bay near the hut (KB, CG, SI), and in forest on the Mt. Prospect track (above Tui Lake) (CG).

Sacred kingfisher (*Todiramphus sanctus*) abundance has diminished. In 2008, CRV did not see any along roads and at their previous forest-edge locations, however CG and SI did observe them along the north coast towards Hutchison Bluff, and on the northern terraces.

Welcome swallow (*Hirundo neoxena*) continue to be present seasonally, many in summer and possibly absent for parts of the winter, but with no indication of nesting. Previously we have attributed this to depauperate invertebrate food sources, but the removal of rats has allowed a notable increase in insect abundance.

Blackbird (*Turdus merula*) appear to have diminished in the denser, darker, forest areas but have increased elsewhere, possibly due to increased food abundance. In 2008 they were notably more abundant on the cleared surface of Boat Cove Road and elsewhere and would give their alarm call before flying well away from the count area. Often their point of departure would be from points close to the counter, but out of sight.

Song thrush (*Turdus philomelos*) rarely called and would fly well away from the track. Their rapid wing-beats were often the only indication of their presence.

Tui (*Prosthemadera novaeseelandiae*) numbers increased with the improving floristic diversity following goat eradication. During the 2008 counts they were very quiet. Whether this was due to the time of year, or whether tui numbers were seriously depleted by a recent spate of mortality observed by Raoul Island staff, is not known. High numbers of dead tui have been observed on at least two occasions in the last few years (K.B. pers obs, 2008, 2009). Necropsies carried out on these birds indicate starvation as a factor in their deaths. Low natural food diversity combined with release from predation pressure and storm events affecting food supplies are possible contributors to these mortality events. Loss of food supplies such as berries and pohutukawa flowers after storm events has been observed (K.B. pers obs, 2008)

Yellowhammer (*Emberiza citrinella*) were relatively common in open areas of goat-browsed vegetation. They diminished in number following goat eradication, but appear to have increased again following rat and cat eradication.

Starling (*Sturnus vulgaris*) were the most abundant bird in the forest in 1967 but declined markedly following goat eradication and forest understory growth. During counts they are often not seen initially but commonly give their short alarm call before flying well away from the count area. DV also repeated counts of starlings flying to roost on the Meyer Islands. This suggested an 80% decline in the number of starlings using that roost. With rats now removed from Raoul it is possible that starlings are learning to roost on Raoul.

Other birds There is no evidence that Tasman masked booby (*Sula dactylatra tasmani*), black noddy (*Anous minutus*), brown noddy (*A. stolidus*), and white-bellied storm petrel (*Fregetta grallaria*) are prospecting Raoul. The former two are commonly seen flying along the shoreline or feeding just offshore, and both breed on the Meyer and Herald Islands.

Fig 3 After the eradication of goats between 1972 and 1985 the forest floor became covered in a dense layer of litter. This photo dated June 1994.

During past visits to Raoul a number of self-introduced passerines have been recorded in low numbers in the forest or at the forest edges. Silvereye (*Zosterops lateralis*), greenfinch (*Carduelis chloris*), goldfinch (*C. carduelis*), and common redpoll (*C. flammea*) were not recorded during the 2008 visit and the condition of the Raoul Island forest leaves little space for them.

DISCUSSION

The forest bird counts were started at a time when we hoped the goats could be removed. Cat eradication had been achieved on small islands and rat eradication was not considered possible. Thus the possible changes of forest condition and bird abundance were not considered in these early data records.

In 1967, we were easily able to count the 100 m wide transect and expected to see, or disturb, all birds on the forest floor and up to the forest canopy. In 1994 and 1998, there was a notable change with the forest floor now being a dense litter layer (Fig. 3), and some fruit-bearing plants increasing in abundance. Total bird numbers had clearly increased, but the density of ground cover may have reduced the opportunity to count some species. By 2008, the forest regeneration had reached a point where many birds were difficult to see. On the Boat Cove Road average visibility was less than 10 metres, both overhead and to the sides, and within that distance (apart from the road surface) the ground cover was sufficient to hide any birds that were on the ground. On the Top Track, average visibility was less than five metres, both overhead and to the sides, and within that distance most of the ground cover was sufficient to hide any birds that were on the ground.

Increased forest density and seasonal changes of bird behaviour meant that in 2008, the most vocal birds were counted far more frequently than the quieter species, and these results are not readily comparable to previous counts.

The eradication of all mammals is allowing a return towards a natural ecosystem, although exactly how and which birds will colonise Raoul is something that only future studies will reveal. Changes of forest bird abundance are similar to those seen in other island ecosystems following the removal of browsing mammals and cats (Diamond and Veitch 1981) but with a further possible influence resulting from increased insect abundance following rodent eradication.

Forest health and management of bird-dispersed invasive plants has reached a level where re-introduction of the New Zealand pigeon can now be considered. Management of the duck population to retain Raoul Island as a grey duck (*Anas superciliosa*) area should also be considered.

With respect to seabirds, black-winged petrels appear to be making a rapid return with multiple colonies becoming established on steep slopes in the drier forested areas; sooty terns have become well-established in the Hutchison's Bluff area (Ismar *et al*. 2010). Red-tailed tropicbirds appear to be nesting on high cliffs, also in the Hutchison's Bluff area. A more gradual return is evident in other species: after six years there is only one confirmed Kermadec petrel breeding record, despite large numbers of the birds on the Meyer Islands and only two white-naped petrels have been seen since the eradications (Gaskin *in press*). Sound broadcast systems should be maintained as the primary method of attracting seabirds back to Raoul Island, but a time limit should be five years from system set up. We recommend that if by 2013 white-naped petrel and Kermadec little shearwater have not established on Raoul Island, chick translocations should be considered.

DOC has established a presence/absence monitoring system for seabirds to be undertaken at the same time as weed eradication work on about 25% of Raoul Island. Monitoring of seabirds, particularly species most at risk and endemic to the region, on all islands in the group on a regular basis is important to understand the health and recovery of the populations of seabirds.

Of equal importance is the requirement to ensure that biosecurity is maintained on all islands in the group. Monitoring for rodents is a primary concern, but there is also potential for introduction of other invasive species.

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Recovery of both a mesopredator and prey in an insular ecosystem after the eradication of rodents: a preliminary study

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Abstract There is growing evidence that rodent eradication often enables the substantial recovery of native species. However, most previous studies have focused on the recovery of conspicuous and charismatic species directly affected by rodents. We examined the responses of the terrestrial ecosystem of Surprise Island, New Caledonia to the eradication of invasive ship rats (*Rattus rattus*) and mice (*Mus musculus*) in 2005. Communities of invertebrates and skinks were compared before and after rodent eradication. Because skinks are prey for rodents and are predators of invertebrates (i.e., mesopredators), we were concerned that rodent eradication would induce mesopredator release (i.e., invertebrates would decrease because of increased skink abundance). Our results showed that skink abundance increased, but counter to our expectations, the abundances of most invertebrates also increased or were not affected. The negative indirect effects of skink abundance on the invertebrate community were likely overcome by both the decreased direct effect of rodent predation and the positive indirect effects of recoveries of other organisms. These included seabirds, which provide resource inputs from the sea and vegetation. These results highlight that increased mesopredator abundance does not always exert negative effects on native ecosystems, and while these changes are important to consider, they should not be the sole reason for renouncing the benefits of eradicating alien predators.

Keywords: Rat, mouse, mesopredator release, indirect effects, top-down effects, bottom-up effects, surprise effect, seabirds, vegetation

INTRODUCTION

Over the past decade, successful eradications of introduced rats (*Rattus* spp.) and mice (*Mus musculus*) have increasingly been reported from islands worldwide (Howald *et al.* 2007). Rodent eradication generally results in the substantial recovery of native species (Towns *et al.* 2006; Howald *et al.* 2007) and is now recognised as a useful restoration tool for island ecosystems (Howald *et al.* 2007). Most previous eradication studies have focused on the recovery of conspicuous and charismatic species such as seabirds and vegetation (Caut *et al.* 2009; Mulder *et al.* 2009). Most of these have examined the direct effects of rodent predation, even though rodents can also affect native species in other ways (e.g., Towns 2009). Therefore, it is important to assess other native groups, such as the invertebrate community, which can have important functions in recipient ecosystems. Moreover, these organisms should be assessed within a community-wide context, as the invertebrate community may not only be affected by direct predation but also by less obvious indirect effects (Fukami *et al.* 2006; Watari *et al.* 2008; Norbury *et al.* 2009; Towns *et al.* 2009). For example, on Amami-Ōshima Island, Japan, the introduced mongoose *Herpestes auropunctatus* has nearly extirpated frogs and skinks by direct predation, resulting in an increase in several insect species that were preyed upon more heavily by frogs and skinks than by the mongoose (Watari *et al.* 2008).

One indirect effect of invasive species eradications can be unexpected population explosions of suppressed species, leading to adverse effects on native ecosystems (Courchamp *et al.* 2003; Zavaleta *et al.* 2001). Examples include introduced mesopredator or herbivore release after invasive predator eradication (Bergstrom *et al.* 2009; Courchamp *et al.* 1999; Rayner *et al.* 2007; Ritchie and Johnson, 2009), and invasive plant explosions after invasive herbivore eradication (Kessler 2001; Kessler 2011; West and Havell 2011). In recent years, such "surprise effects" have raised awareness of the importance of long-term monitoring and an ecosystem-wide perspective during eradication efforts (Simberloff 2001). However, studies that consider these factors are rare.

In the present study, we examined the preliminary results of a long-term project on Surprise Island, New Caledonia. We eradicated the ship rat (*Rattus rattus*) and mouse population on this island by poisoning in 2005 and monitored the entire ecosystem, specifically targeting seabirds, sea turtles, lizards , invertebrates, and vegetation before and after the eradication (Caut *et al.* 2009; Courchamp *et al.* 2011). Rodents can affect lizard populations as well as the invertebrate community (Towns *et al.* 2006). Our preliminary analysis of stomach contents of skinks on Surprise Island indicated that skinks prey on terrestrial invertebrates such as insects, spiders, isopods, and land snails (Watari *et al.* unpublished data). We thus expected that the eradication of rodents would be followed by an increase in the abundance of skinks (a mesopredator) with a concomitant decline in the mesopredator's prey of terrestrial invertebrates, whereas there would be no effect on flying insects that are less vulnerable to skink predation. We analysed the results of skink and invertebrate abundances, with special attention to this potential "surprise effect".

MATERIALS AND METHODS

Study site

Surprise Island (Fig. 1), on the D'Entrecasteaux Reefs 230 km north of the main island of New Caledonia, is 24 ha in area and reaches 9 m elevation. Habitats on the island include a central open patch (the "plain") with bare ground and patches of various herbaceous plant species (e.g., Graminae, Compositae, and Portulaceae) surrounded by woody vegetation dominated by *Argusia argentea* Heine, *Suriana maritima* Arnott, *Scaevola sericea* Gaertn and *Pisonia grandis* Brown (Caut *et al.* 2008, 2009; Fig. 1).

Surprise Island provides refuge for 14 species of seabird, 10 of which breed on the island. Ship rats and house mice were probably introduced to Surprise Island during guano mining in the late 19th to the early 20th century, and/or in the late 20th century, when an automatic meteorological station was established. Two species of terrestrial reptiles were also likely introduced to the island: a New Caledonian

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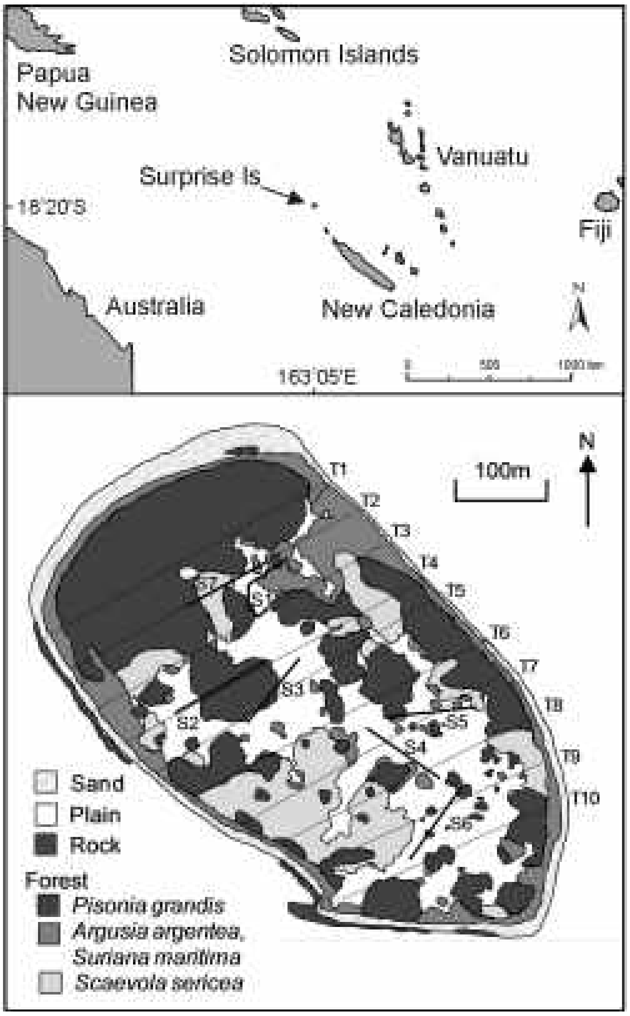

Fig. 1 Surprise Island and its four major distinct habitats of open sand flat and three vegetation types (modified from Caut *et al.* 2009). T1–T10 indicate invertebrate transects. S1–S7 indicate skink transects.

skink (*Caledoniscincus haplorhinus*), and a non-native gecko (*Lepidodactylus lugubris*) (Caut *et al.* 2009).

Rats and mice are assumed to have been eradicated, as none have been detected since 2006 following the application of rodenticide in 2005, despite trapping and hair trap surveys for four years (Caut *et al.* 2009, Courchamp *et al.* unpublished data).

Assessing animal communities

We compared the community composition four years before rodent eradication (2002–2005 for the skink and 2003 for invertebrates) and four years after the eradication (2006–2009 for both the skink and invertebrates). Surveys were conducted in November and December of each year (Caut *et al.* 2009).

For the estimates of skink abundance, we established seven 100 m transects in the main habitat unit (Fig. 1), along which we counted the number of skinks within a 2 m width during 15 minute walks. The transects were located in the plain and *Pisonia grandis* patches, as the dense vegetation in the other forest patches, such as *Augusia argentea*, *Suriana maritima*, and *Scaevola sericea*, made it difficult to conduct lizard surveys (Caut *et al.* 2009; Fig. 1). Surveys were conducted between 12:00–15:00 hours on three separate days per visit. As some transects traversed plain and forest vegetation types, each transect was also divided into four 25 m-long sub-transects, for which the number of skinks and major vegetation types were recorded. We also recorded the weather conditions (sunny or not sunny), which were likely to affect skink activity.

To collect invertebrate samples, we used yellow surface traps (20 × 20 × 10 cm) primarily for flying insects as well as pitfall traps (10 cm diameter × 15 cm height) mainly for ground-dwelling invertebrates. All traps were partially filled with soapy water and set along the 10 transects across the island, spaced 50 m apart to maintain independence between traps (Fig. 1). Together, the transects covered a total of about 3000 m, covering all habitats on the island. Arthropod traps were deployed one time per visit over 48 h in 2003 and 2006 and over 24 h in 2007–2009 (surface traps every 75 m and pitfall traps every 50 m; Fig. 1). Trapped invertebrates were stored in 70% alcohol until identification in the laboratory. We analysed data from 20 surface traps and 29 pitfall traps from 2003 (before eradication) and 38 surface traps and 60 pitfall traps from 2006–2009 (after eradication). Invertebrates with lengths >3 mm were assume to be in the size range of skink prey and were included in analyses, but ant samples were excluded, as a separate analysis was conducted for ant populations (Cerda pers. comm.).

Statistical analyses

To examine the effect of rodent eradication on the skink population, we used a generalised linear mixed-effect model (GLMM) with Poisson distribution (Faraway, 2006) using R (R Development Core Team, 2007) with the lme4 package (Douglas 2007). We used the number of skinks observed along each 25-m sub-transect (*Skink*). Because there may be a time lag for numerical responses of skinks to dynamics of the rodent populations (Schmidt and Ostfeld 2003), we assumed either no delay, a 1-year delay, or a 2-year delay to the effect of rodent eradication. Presence and absence of rodents was assigned values of 0, 1 in either the year of ($Eradication_{year+0}$), 1 year before ($Eradication_{year-1}$), or 2 years before ($Eradication_{year-2}$) the actual skink field surveys as explanatory variables. The effect of vegetation type (*Vegetation*; forest and plain: 0, 1), and their interaction (*Eradication* × *Vegetation*) were also included in the model as fixed factors, because the strength of top-down effects may vary in different environments (Towns *et al.* 2003; Rayner *et al.* 2007; Ritchie and Johnson 2009). We also included *survey-day*, *survey-year*, *transect*, *sub-transect*, and *weather* (sunny or non-sunny) as random factors, all of which may affect the number of observed skinks. Based on Akaike's Information Criterion (AIC) values, we conducted model selection among the models with all possible combinations of factors.

To examine the effects of rodent eradication on the community composition of invertebrates, we conducted two separate redundancy analyses (RDAs) for the samples caught in surface and pitfall traps. In these analyses, the capture rate of each species per trap-night was used as the response variable, and both *Eradication* (before and after rodent eradication: 0, 1) and *Vegetation* (forest and plain: 0, 1) were included as explanatory variables. Among the three types of forest patches (Caut *et al.* 2009; Fig. 1), we analysed data from the patches of *Pisonia grandis* in the preliminary study. Unfortunately, because we lack replication in the year before eradication (i.e., we only have before-eradication data from 2003), we did not consider the effect of the survey year. In the RDAs, the significance of each explanatory variable was tested using comparisons to Monte Carlo permutations with 999 iterations. All RDAs and permutation tests were performed using CANOCO for Windows, version 4.5.

To illustrate the patterns of the response of each species to rodent eradication (when *Eradication* was detected as a significant factor), another RDA was conducted using *Eradication* and *Vegetation* as fixed and random factors, respectively, from which species scores on the first axis could be considered characteristics of species response to rodent eradication (Leps and Smilauer 2003).

RESULTS

Skink population

The abundances of skinks observed in the transect surveys from 2002 to 2009 increased substantially after rodent eradication (Fig. 2). The model 1 with $Eradication_{year\text{-}1}$, *Vegetation*, and $Eradication_{year\text{-}1}$ × *Vegetation* as the explanatory variables was clearly superior to the other models (ΔAIC of all the other models > 2) (Table 1).

Invertebrate community

We collected at least 40 taxa of invertebrates in surface traps and 35 species in pitfall traps, covering a total of 13 orders (Table 2). The surface traps more frequently captured a greater diversity of invertebrates (e.g., Diptera, Hemiptera, and Hymenoptera) than did the pitfall traps (Table 2). Based on the Monte Carlo permutation tests of RDA ordinations (Fig. 3), an effect of rat and mouse eradication was not detected in the invertebrate community caught in surface traps but was significant among invertebrates collected in pitfall traps (surface trap: λ = 0.019, F = 1.093, P = 0.316, Fig. 3a; pitfall traps: λ = 0.094, F = 9.126, P = 0.001, Fig. 3b). The effect of vegetation on invertebrate community composition was significant in surface traps and was marginal in pitfall traps (surface trap: λ = 0.047, F = 2.791, P = 0.018, Fig. 3a; pitfall traps: λ = 0.026, F = 2.555, P = 0.062, Fig. 3b). The RDA ordination diagram with vegetation type as a random (v fixed) factor is presented in Figure 3c. RDA scores of the 12 species with sufficient sample sizes (frequency of occurrence > 0.1; Table 2) obtained from Fig. 3c and summarised in Fig. 4 identified nine species with negative coefficients (i.e., directing toward the *Eradication* axis in Fig. 3c), indicating that they were positively affected by rodent eradication.

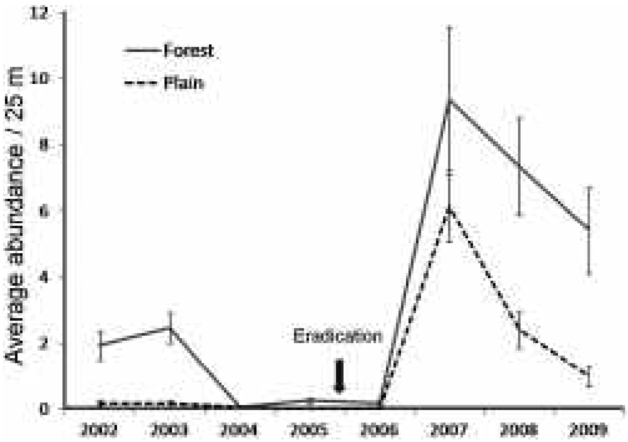

Fig. 2 Average abundance of skinks (± SE) observed in 25 m sub-transects.

Table 1 The GLMM models explaining skink abundance and their AIC values. All models include *survey-day*, *survey-year*, *transect*, *sub-transect*, and *weather* as random factors.

Model	Combination of explanatory Variables	Estimate	SE	AIC	ΔAIC	deviance
1	$Eradication_{year\text{-}1}$	1.9536	0.7994	1011	-	992.8
	Vegetation	-2.0106	0.3130			
	$Eradication_{year\text{-}1}$ × *Vegetation*	1.6228	0.2411			
2	$Eradication_{year\text{-}0}$	0.8036	1.0695	1020	9	1002
	Vegetation	-1.9732	0.3135			
	$Eradication_{year\text{-}0}$ × *Vegetation*	1.5800	0.2416			
3	$Eradication_{year\text{-}2}$	2.3085	1.1077	1043	32	1025
	Vegetation	-0.2475	0.2210			
	$Eradication_{year\text{-}2}$ × *Vegetation*	-0.6608	0.1207			
4	$Eradication_{year\text{-}1}$	2.4517	0.7982	1068	57	1052
	Vegetation	-0.5170	0.2151			
5	$Eradication_{year\text{-}1}$	2.4479	0.7991	1071	60	1057
6	*Vegetation*	-0.5159	0.2152	1072	61	1058
7	$Eradication_{year\text{-}2}$	2.0187	1.1074	1072	61	1056
	Vegetation	-0.5163	0.2152			
8	$Eradication_{year\text{-}0}$	1.2927	1.0682	1073	62	1057
	Vegetation	-0.5161	0.2152			
9	*Null Model*	-	-	1075	64	1063
10	$Eradication_{year\text{-}2}$	2.016	1.108	1075	64	1061
11	$Eradication_{year\text{-}0}$	1.290	1.069	1076	65	1058

Table 2 Frequency of occurrence of invertebrate species per trap (FO).

Species	ID*	FO in surface traps[1]	FO in pitfall traps[2]
BLATTODEA			
Blaberidae sp.	1	0.052	0.017
Blattidae sp.	2	0.181	0.067
COLEOPTERA			
Coleoptera sp.1	3	-	0.044
Chrysomelidae sp.	4	0.034	-
Coccinellidae sp.	5	0.009	-
Curculionidae spp.	6	0.241	0.111
Tenebrionidae sp.	7	0.009	0.244
Coleoptera sp.2	8	-	0.006
Coleoptera sp.3	9	-	0.011
DERMAPTERA			
Forficulidae sp.	10	-	0.206
DIPTERA			
Drosophilidae sp.	11	0.017	0.006
Asilidae sp.	12	0.043	-
Stratiomyidae sp.	13	0.034	-
Tachinidae sp.	14	0.103	0.006
Diptera sp.	15	-	0.011
Pipunclidae sp.	16	0.19	-
Dolichopodidae sp.	17	0.396	-
Therevidae sp.	18	0.017	-
EMBIIDINA			
Oligotomidae sp.	19	0.052	0.117
HEMIPTERA			
Anthocoridae sp.	20	0.112	0.011
Cicadellidae spp.	21	2.043	0.628
Delphacidae spp.	22	0.034	-
Eurymelidae sp.	23	0.043	0.028
Cydnidae sp.	24	-	0.033
Hemiptera sp.1	25	0.052	-
Hemiptera sp.2	26	0.043	-
Hemiptera sp.3	27	0.121	0.044
HYMENOPTERA			
Ichneumonidae sp.	28	0.026	-
Halictidae sp.	29	0.026	-
Platygasteridae sp.	30	0.155	-
Pteromalidae sp.	31	-	0.011
LEPIDOPTERA			
Geometridae sp.	32	0.052	0.006
Lepidoptera sp.1	33	0.034	0.011
Lepidoptera sp.2	34	2.095	0.128
Lepidoptera sp.3 larvae	35	0.043	0.811
Sphingidae sp.	36	0.328	0.011
Lepidoptera sp.4	37	-	0.006
Sphingidae sp. larvae	38	0.017	-
Lepidoptera sp.5	39	0.052	-
Lepidoptera sp.6	40	0.017	-
ORTHOPTERA			
Acrididae sp.	41	0.078	0.083
Gryllidae sp.	42	0.164	0.006
Mogoplistidae sp.	43	0.052	-
ISOPODA			
Isopoda sp.1	44	-	0.4
Isopoda sp.2	45	0.5	14.38
Armadillidae sp.	46	0.017	1.767
ARANEAE			
Araneae sp.1	47	0.069	0.106
Heteropodidae sp.	48	-	0.022
Araneae sp.2	49	0.026	-
Lycosidae sp.	50	-	0.017
Araneidae sp.	51	-	0.011
PULMONATA			
Pulmonata sp.	52	-	0.194
DECAPODA			
Paguroidea	53	-	0.022
Unidentified larvae	54	0.06	-

*: IDs are used for Fig. 3a, b, c

DISCUSSION

Our results indicate that rodent eradication positively affected populations of skinks and terrestrial invertebrates, but did not affect flying insects. However, a closer examination of the data provides a slightly more complex picture.

The GLMM model with $Eradication_{year-1}$ as the response variable was selected as the best model, indicating that the response of skinks to eradication was observed with a 1-year time lag. Similar delayed responses to predator abundance have been reported for songbirds with varying predator pressure on eggs and chicks (Schmidt and Ostfeld 2003). Rodents may also primarily consume the eggs or juveniles of the skink, leading to the observed delayed response, although we lack observations of such events. There are some indications that tuatara (*Sphenodon punctatus*), an endemic reptile of New Zealand, is suppressed by the rats through predation of eggs or juveniles (Towns *et al.* 2006), although the tuatara is considerably larger than *C. haplorhinus*. Another possibility is that recovery of the skink population lagged behind recovery of its food or habitat (i.e., invertebrates and vegetation). Further studies are needed to reveal the above processes. Although skink counts seemed to decrease after 2007 (Fig. 2), this might not have been caused by a decrease in the skink population, but by changes in skink detectability because of weather conditions. To test for this, we incorporated weather into the GLMM as a random factor. There were 2 and 1 days with non-sunny weather during the three surveys in 2007 and 2008, respectively; more skinks were observed on these days than on sunny days. Indeed, the average (± SE) numbers of skinks per 25-m sub-transect in 2007–2008 under sunny and non-sunny days were 3.67 (± 0.67) and 13.0 (± 2.20) in forest patches, and 0.88 (± 0.24) and 7.80 (± 1.10) in plain patches.

These results indicate that skinks were observed more frequently in forest patches and that their abundance increased in both forest and plain patches after rodent

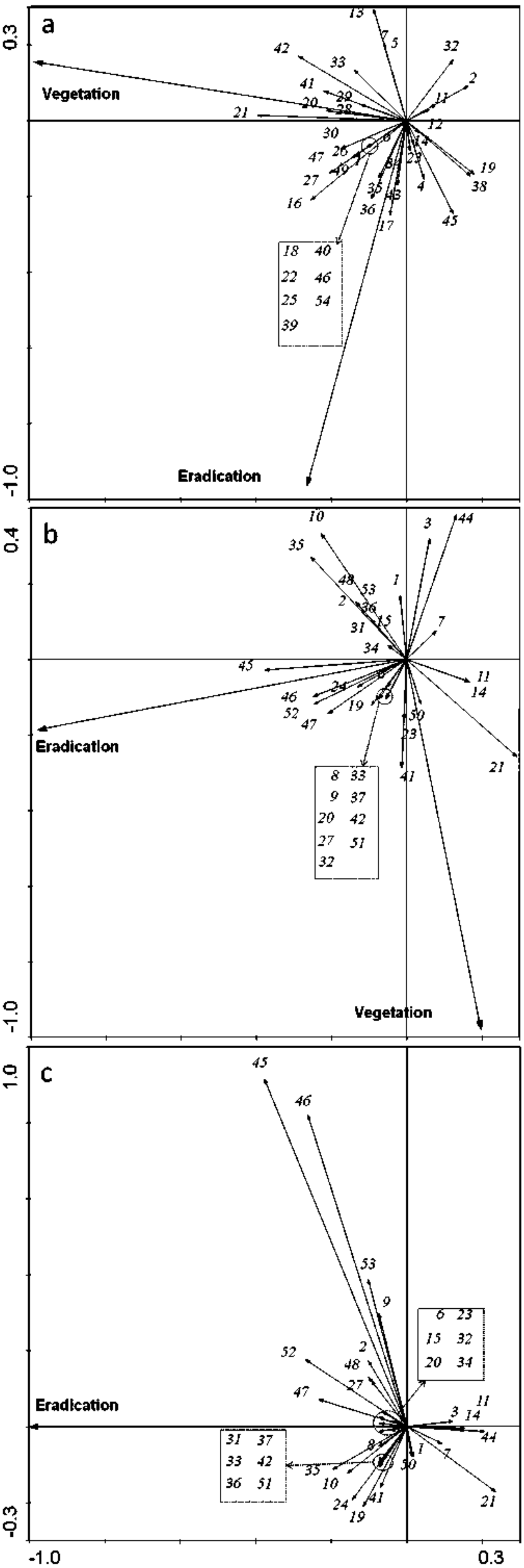

Fig. 3 RDA ordination diagrams of invertebrate community caught by a) surface traps, b) pitfall traps, and c) pitfall traps with the effect of vegetation type as a random factor. Numbers represent the ID of each species from Table 2. The horizontal and vertical axes are the first and the second RDA axes respectively. Species arrows directing toward the *Vegetation* and *Eradication* show that the species frequently occurred at the plain (vs. forest) and after eradication (vs. before eradication), respectively. For example, Fig. 3b indicates that species 21 occurs more frequently either at the plain patches or before eradication.

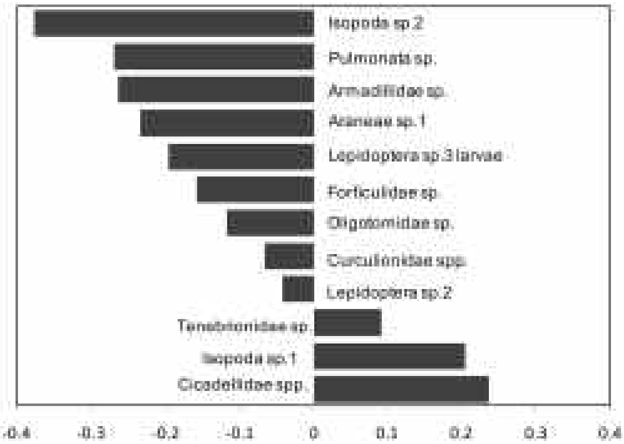

Fig. 4 RDA scores of 12 major species plotted against the first (horizontal) axis of Fig. 3c. Species with positive and negative RDA scores (i.e. species arrows directing away from, and toward the *Eradication* in Fig. 3c) indicate decreasing and increasing patterns following rodent eradication.

eradication. However, the extent to which skinks increased depended on vegetation type, with a greater increase in plain patches than in forest patches. This pattern can be explained by the positive relationship between skink abundance and vegetation ground cover (Norbury *et al.* 2009). Although ground vegetation at the study site has recovered since rodent eradication in both forest and plain patches (Courchamp *et al.* 2011, unpublished data), vegetation cover has also increased in areas that were once bare ground (Courchamp *et al.* unpublished data), likely leading to stronger bottom-up effects on the skink population.

We did not observe our predicted "surprise effects," where invertebrate abundance declined after rodent eradication, despite the expected increase in skink (mesopredator) abundance. In fact, ground-dwelling invertebrates from pitfall traps increased in abundance after rodent eradication, whereas the flying insects in surface traps showed a neutral response to the eradication. Therefore, the invertebrate communities generally benefited from the removal of their top predators (rodents), despite the increased abundance of their mesopredator (skinks). To thoroughly examine the effects of rodent eradication, we compared invertebrate community structure between years before and after rodent eradication. However, caution is required when interpreting such differences in invertebrate community composition, as they are likely to be caused not only by a balance of top-down predation effects between rodents and skinks, but also by other indirect effects. Furthermore, vegetation (food and habitat resource) and seabirds (resource input from the sea) have clearly recovered since the rodent eradication on Surprise Island (Courchamp *et al.* 2011, unpublished data), which could, in turn, exert bottom-up effects on the invertebrate community (Fig. 5; Fukami *et al.* 2006; Norbury *et al.* 2009; Towns *et al.* 2009). How these responses might be induced requires further examination. In summary, any negative indirect effects of increased skink abundance (mesopredator increase) on the invertebrate community were likely overcome by the sum of the decreased direct effect of rodent predation and the positive indirect effects of the recoveries of seabirds with increased nutrient input and vegetation through decreased rodent consumption (Fig. 5). Moreover, the increase in invertebrate abundance may partially contribute to the increased skink abundance through a bottom-up cascade. We thus found that increased mesopredator abundance does not always exert negative impacts on the rest of the community, and while important to consider, should not be the sole reason for renouncing the benefits of eradicating alien predators (Bonnaud *et al.* 2010; Russell *et al.* 2009).

The responses of some invertebrates may reflect interspecies interactions within the invertebrate community. Among the three Isopoda species sampled in this study, Isopoda sp.2 showed the highest rate of recovery and Armadillidae showed the third-highest rate. Another Isopoda species, Isopoda sp.1, showed a negative response. A possible explanation for these different responses between species with similar traits is that these patterns were the result of competition among them. Two predatory invertebrates, the spiders Araneae sp.1 and Forficulidae sp., both became more common in our samples. These increases might have been caused by a reduction in top-down pressure and increased bottom-up effects through increases in other invertebrates.

Because we only analysed the abundances of skinks and invertebrates in this study, the relative contributions of possible mechanistic processes to the observed patterns in Fig. 5 remain unknown. Our next challenge will be to analyse the strengths of interactions in light of predator and prey densities, quantitative food habits, species traits, and interactions within invertebrate communities. Our study lacks replication because we only examined one island, and three nearby islands are ecologically very different. In addition, we only had invertebrate samples from 1 year before eradication. We thus cannot exclude the possibility that the above patterns resulted from factors other than the rodent eradication, such as annual climate variation, although the skink and invertebrate recoveries shown in this study are consistent with the results of other studies (e.g., Towns *et al.* 2006). Moreover, the lack of any information about ecosystem structure before rat and mice introductions to this island makes it difficult to assess the extent to which changes within communities after rodent eradication represent a recovery towards the initial state. Our study of Surprise Island communities after alien rodent eradications also reveals the difficulty of adequately understanding ecosystem processes, even in apparently very simple, small closed ecosystems. We must continue to carefully monitor the Surprise Island ecosystem. Nonetheless, our results and conclusions are important both ecologically and in terms of conservation efforts, particularly for highlighting some limitations of ecosystem studies.

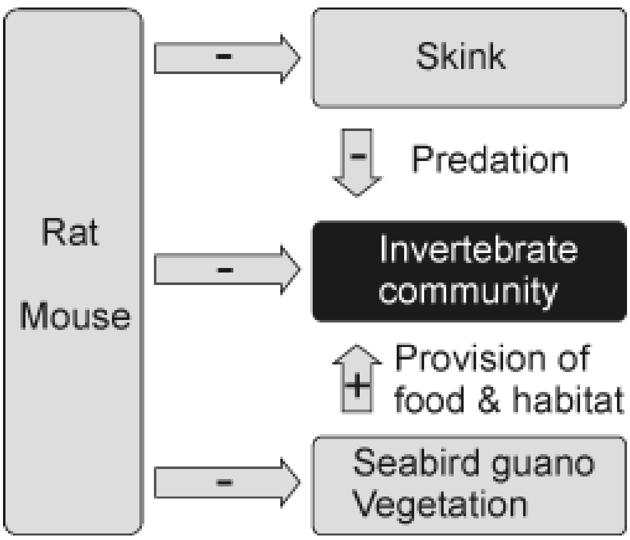

Fig. 5 Possible ecosystem processes related to rodent eradication and the invertebrate community.

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People, Policy, and Prevention

The roles of, and approaches to, eradication operations that involve people, policy makers, and then biosecurity measures to prevent future alien species invasions.

Eradications of invasive mammals on islands in Mexico: the roles of history and the collaboration between government agencies, local communities and a non-government organisation

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Abstract Eradications of invasive mammals have over the last decade been a key element in the restoration of Mexican islands. To date, 48 eradications have been completed on 30 islands. This work has provided the climate for a wider movement towards the protection and restoration of Mexican islands involving many players and institutions. Perceptions of islands have changed from earlier abuse and abandonment to current realisation of their importance for sovereignty, their rich biodiversity, and their potential as sites for sustainable development. This increased awareness was followed by social acceptance of the importance of islands, organisational development of advocates for them, scientific research, secure funding for projects on them, and institutional support for this work. A collaborative network now includes federal government agencies such as the Mexican Navy alongside academic institutes and universities, local communities, artisanal fishermen co-operatives, non-government organisations (NGOs), and national and international donors. A crucial NGO component of this network has been the Grupo de Ecología y Conservación de Islas, A.C. (GECI). This private organisation now has a field staff of 24 scientists and technicians who work closely with personnel from government agencies. Soon all Mexican islands will be legally protected under federal categories. Permanent government staff are being recruited, and activities on islands are starting to be supported with federal budgets. These public budgets may also soon supplement funds provided by private donors. During the last decade, most funds have been provided by American private foundations; comparatively Mexican private and public funding is still limited. There have been positive outcomes from international collaboration and exchange. If the current pace is sustained a strategic goal could be met: to eradicate all invasive mammals from the remaining Mexican islands by the year 2025.

Keywords: Non-government organisation, invasive species, island conservation, Mexico, islands biodiversity, restoration

INTRODUCTION

A disproportionate number of global extinctions have been on islands, often as a direct result of invasive species (Veitch and Clout 2002). In response to this, invasive species of mammals have been removed from numerous islands in many locations. The rate of removal has been particularly high in Mexico, where 48 successful eradications of large mammals and rodents have been conducted on 30 islands (Aguirre-Muñoz *et al.* 2011). In this paper, we describe how the relationship of Mexico with its insular territories has changed over time and how restoration activities have developed. We use an interdisciplinary approach that integrates the perspectives of environmental conservation and interpretative sociology. Our aim is to identify how the historic, social, institutional, organisational and financial contexts developed so that invasive species could be eradicated from islands. Specific questions include: What have been the directions and the intentions of the diverse 'social actors' towards Mexican islands? Are there relevant historic changes in the relationship of Mexico with its islands? What factors contributed to these changes? Did island restoration activities such as the removal of invasive species contribute? And finally, how have the successful eradications been conducted?

Biodiversity protection, sovereignty, and sustainable development are the three axes used for the analysis. There may be potential biases posed by the authors' active involvement in island conservation and natural resources use issues in Mexico. However, this can also be viewed as providing the richness of an insiders' experience, assuring rapport between the analyst and the research subject (Russell Bernard 2006).

Within the framework used here, 'social actors' include individuals or collectives, including fishermen's organisations, government officers or agencies, civil society groups, researchers and academic institutions. They all actively and consciously interact with the changing world around them as well as with other social groups, and with historic consciousness of their own acts (Long and Long 1992; Touraine 1969, 1987). The concept of 'social actor' implies that individuals and organisations have the capability to comprehend their own social experiences and can effectively respond to the challenges posed by their everyday life and current contingencies, envisioning alternatives to improve their future and implementing these. Understanding the intention and direction given by the actors to their actions are central to comprehensive sociology, representing its very methodological foundations (Weber 1984).

We begin with a historical account of how the protection of the Mexican islands unfolded, and the roles played by diverse actors in this process, but particularly the part played by a non-government organisation (NGO). We also show how, because of its ecological importance, effectiveness, and success, the eradication of invasive mammals has helped to develop a new paradigm for Mexican islands, characterised by strong protection and innovative conservation actions.

It may be inappropriate to suppose that this successful story can act as a model for other regions or countries, because every country or region has its own and history, and particular cultural, social, or economic setting. However, there might be parallels between this "Mexican case" and the development of conservation ethics, practices and organisations elsewhere. This is particularly true for the role of NGOs (e.g., Wilson 2002), a point that we will return to later. Before doing so, we describe how the scene was set for raised awareness, and how this was followed by social acceptance, knowledge, infrastructural support, funding, and, finally, the institutional processes that have now started to support the achievements.

Pages 386-394 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

HISTORY OF THE MEXICAN VIEW OF ISLANDS

Greed, abandonment, and weakness

Before the first Spanish contact, some Mexican islands were inhabited or visited by pre-Hispanic native groups. Those visited included Cedros, off the Pacific coast; some islands of the Gulf of California (Bahre and Bourillón 2002); and Mujeres and Cozumel, in the Caribbean. On Cedros Island, the native "Cochimíes" developed a distinct marine culture (Del Barco 1988). However, most Mexican islands are arid and lack fresh water. The oceanic islands far from the continent, such as the Revillagigedo Archipelago and Guadalupe Island (Fig 1), were not even visited by Native Americans.

During the European discovery of the American continent and the early conquest of Mexico, conquerors competed intensely to claim as much new territory as possible. Because islands have strategic value for navigation and military purposes they received particular attention. Mythical views also permeated this interest as is demonstrated by the historic origin of the "California Island" that was described in fiction well before the name was assigned to a real location. The word California first emerged in the late 15th century, when descriptions of a utopian island appeared in the classic Spanish cavalry book *"Las Sergas de Esplandián"* (The heroic adventures of Esplandián). Esplandián, the heroic fiction character was the first son of the Spaniard Amadís de Gaula and a Great Britain Princess. The book was originally published in 1490 (Rodríguez de Montalvo 1526) as part of a series of Spanish romances that were very popular in Europe. Among the places visited by Esplandián, the book states:

"Know that on the right hand of the Indies there was an island named California, very close to Earth's Paradise, inhabited only by black women, with no single male in there…, women rich in pearls and gold …".

This wishful thinking became a reality when the mythical "island" of California was discovered and named by the Spanish conquerors.

Another symbolic view forms an essential part of Mexican historic identity. The national seal represents an image revealed to an Aztec priest that headed an epic diaspora from a coastal island on Mexico's northwest to the current valley that hosts Mexico City (Enciclopedia de México 1987). The divinity indicating the end of their journey would be an eagle devouring a snake. The eagle and the snake represent a fusion of complementary symbolic forces. The eagle represents the day, the sun and the diurnal sky; the snake symbolizes the night, the moon and the nocturnal sky. The Aztecs' Promised Land was found: a fertile valley with a lake; in the middle of the lake, an islet with a cactus tree; on top of the cactus tree, a golden eagle devouring the snake (Fig. 2). These elements are in the Mexican National seal. The islet, at the core of the founding Aztec territory became Mexico City, the geographical and political centre of the current country.

Fig. 2 The Aztec's founding myth of the Promised Land (an island), now Mexico City. "La Fundación de México". Colour lithograph by J.G. Posada, 1900.

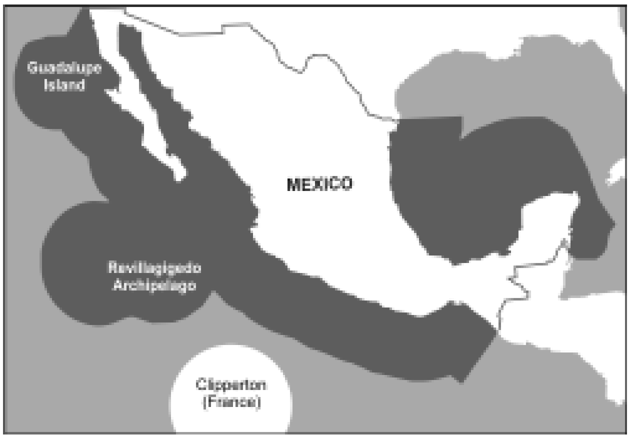

Fig. 1 Mexico and its Exclusive Economic Zone (EEZ), which at 3,149,920 km^2 is the 13th largest in the world, and larger than its terrestrial territory of 1,964,375 km^2.

When Spain permanently departed the Americas, and with the independence of Mexico early in the 19th century, the colony's vast maritime power was lost. Islands were not a priority for the new country. In order to confer some legal protection, Mexican islands were decreed as federal territories by successive constitutions and remained so in the Constitution of 1917, at the birth of the modern country after the Mexican Revolution. During the 19th and early 20th centuries, leases to exploit guano on several islands were granted to private companies, some linked to foreign interests (González Avelar 1992). As part of that period and following an international dispute, France gained possession of Clipperton Island in the tropical Pacific Ocean off Acapulco (Fig 1); the island is still a French possession (González Avelar 1992; Restrepo 1999).

Sovereignty and natural resources

These experiences encouraged modern Mexican authorities to increase their presence and sovereignty over the islands. In order to induce settlement and exercise sovereignty on the (then remote) Baja California Peninsula and nearby islands, fishermen cooperatives were given financial and technical assistance, and received long-term and exclusive fishing rights to abalone and lobster.

In 1983, Mexico signed the United Nations Convention on the Law of the Sea (UN 1982). Linked to the international adoption of the Exclusive Economic Zone (EEZ), Mexico developed military and productive infrastructure and establish permanent settlements on its remote islands, as a means of exercising granted sovereign rights on the islands and the EEZ (Fig. 1). Permanent Navy facilities, garrisons, piers and airfields were built on the remote Socorro, Clarión, and Guadalupe islands. Other islands, closer to the mainland, have permanent Navy facilities, and have permanent fishing villages.

ISLAND CONSERVATION

Raised awareness: the early years

Protection of the ecological integrity and natural resources of islands dates back to 1922, with a presidential decree to protect the wildlife of Guadalupe Island and its surrounding waters (DOF 1922). By that time, the Guadalupe Island fur seal (*Arctocephalus townsendi*) and the Northern elephant seal (*Mirounga angustirostris*) populations had been overexploited and were at risk of extinction.

Except for such rare cases, the relevance of biodiversity, conservation of insular ecosystems, and sustainable use of fisheries only became apparent in Mexico during the second half of the 20th century. Movement towards environmental conservation and wise use of the natural resources gained momentum in Mexico over the past three decades, accompanying similar global views. The first interest in the ecology and conservation of Mexican islands came from academia, with the pioneering comprehensive compilation on the ecology of Mexican islands published by Case and Cody (1983). The first applied island conservation actions were combined with scientific research, when the National Autonomous University (UNAM) initiated one of the first successful island conservation projects. The eradication of invasive mammals started in 1994 with the removal of house mouse (*Mus musculus*) and ship rats (*Rattus rattus*) from Rasa Island, a seabird sanctuary in the Gulf of California (Tershy 1995; Bahre and Bourillón 2002). Soon thereafter the first comprehensive review on the Gulf of California Islands was undertaken (Bourillón *et al*. 1988) and the first Official Atlas of Mexican islands (INEGI 1990) was published.

Following this early conservation activity, a small bi-national group of US-Mexican biologists conceived the possibility of restoring northwest Mexican and US islands by eradicating invasive vertebrates (Bernie Tershy and José Á. Sánchez-Pacheco pers. comm.). Two private NGOs were established by the end of the 1990s to assist with this: one in the US (Island Conservation; hosted by the University of California in Santa Cruz) and one in Mexico (Grupo de Ecología y Conservación de Islas; GECI). By the early 2000s, these two organisations had successfully collaborated over the eradication of several species of invasive animals on islands of both countries (Aguirre *et al*. 2008; Samaniego-Herrera *et al*. 2009; Tershy *et al*. 2002; Wood *et al*. 2002). After 2002, the Mexican organisation started to unfold on its own, became autonomous, and has developed working relationships inside Mexico as well as collaborative links with teams dealing with invasive species elsewhere, including New Zealand, Australia, USA, Ecuador, Canada, Cuba, Brazil, Argentina, Chile, Dominican Republic, and with international organisations.

Enduring, successful and tangible results of invasive species eradications on Mexican islands during the last decade (see Aguirre-Muñoz *et al*. 2011) have attracted attention from government, local communities, fishermen organisations, donors and academic institutions. Coupled with greater understanding of biodiversity on the islands, the successful eradications have sown the seeds of a wider movement, with impacts and concerns beyond the scope of eradications. Two threads have since emerged. One views islands as ecologically valuable territory, integrating them with issues of sovereignty and sustainable development. The second builds on the introduced species issue as the basis for a new perspective of Mexico's mainland territory. A recent dispute over the use of Coronado Sur Island illustrates the former.

Development of a social movement for conservation

Island conservation reached a complex array of actors and institutions in Mexico as a result of conflict over Coronado Sur Island, adjacent to the border between the USA and Mexico. The conflict did not originate from local communities but came as a result of globalisation. Tensions developed when the multinational petrochemical company ChevronTexaco proposed building a liquefied natural gas (LNG) regasification facility adjacent to Coronado Sur Island. The gas would then cross the border by a pipeline to San Diego, USA.

The Coronado archipelago contains four species of endemic reptiles, two subspecies of endemic terrestrial birds and one species of endemic rodent. Pinnipeds and seabirds are abundant. The vegetation of Coronado Sur has not been heavily modified and the island supports the world's largest population of a subspecies of Xantus' murrelet (*Synthliborampus hypoleucus scrippsi*), which is a listed threatened species in Mexico and the USA. Feral donkeys (*Equus africanus*) and goats (*Capra hircus*) were removed from the island, although house mice are still present. Local fishermen harvest abalone (*Haliotis* spp.), lobster (Palinuridae), and sea urchin (Echinodermata) from around Coronado Sur Island and the northernmost Mexican Navy base is on the island.

Before the LNG project started, a proposal was presented in 2003 by GECI and the Protected Areas Commission (Aguirre Muñoz *et al*. 2003) to protect the Baja California Pacific Islands. The initiative was backed by the Mexican Congress of the Union, which then passed a resolution requesting the Federal Government to publish the protection decree, to eradicate the invasive pests on the region's islands (Congreso de la Unión 2003, 2007), and to confer Biosphere Reserve status over all the islands

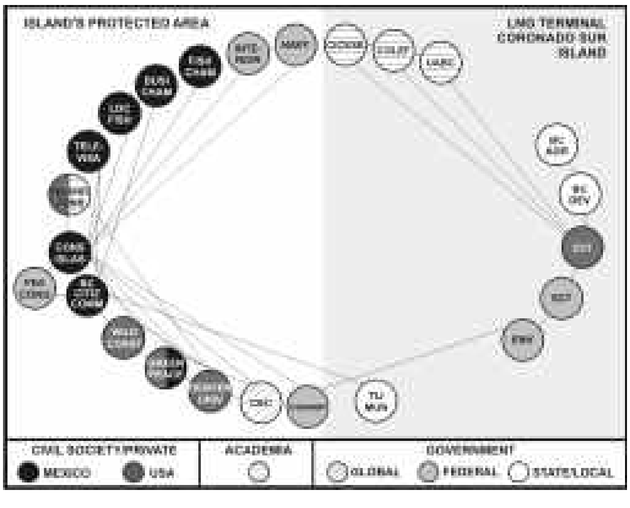

Fig. 3 General sociogram showing the social actors and agencies choosing between the LNG facility on Coronado Sur Island and a new protected area. The lines represent formal or informal linkages between the involved actors or agencies.

Table 1 Participants in debate over the use of Coronado Sur Island, Mexico, identified in the general sociogram (clockwise), with intensity of involvement identified.

Actor/Agency (Acronym)	Full name	Involvement	Intensity
CICESE	Centro de Investigación Científica y Educación Superior de Ensenada	Federal Government Research Centre. Contracts from ChT.	Low
COLEF	Colegio de la Frontera Norte	Federal Government Research Centre. Contracts from ChT.	Low
UABC	Universidad Autónoma de Baja California	State University. Contracts from ChT.	Low
BC AGR	Baja California State Agriculture and Fisheries Promotion Ministry	Baja California State Government (vs. Protected Area)	High
BC DEV	Baja California State Development Ministry	Pro LNG - Baja California State Government	High
ChT	ChevronTexaco de México, S.A. de C.V.	Pro LNG - Project developers	High
SCT	Communication and Transportation Ministry, Federal Government.	Pro LNG - Lease to ChT	High
ENV	Environmental Ministry, Federal Government.	Pro LNG - Lease to ChT EIA approval.	High
TIJ MUN	Municipality of Tijuana	Pro LNG - Permit for the LNG pipeline	Medium
CONANP	Natural Protected Areas Commission, Federal Government	Pro Protected Area – Promotion	High
CEC	Commission for Environmental Cooperation (Canada, US and Mexico), NAFTA	Pro Protected Area - Environmental law compliance review	Medium
DENVER UNIV	Denver University, USA.	Pro Protected Area - Integration of the citizen's petition to the CEC	High
GREENPEACE	Greenpeace Mexico and Greenpeace USA	Pro Protected Area - Protests and public opinion campaigns	High
WILDCOAST	US (California) NGO	Public opinion campaigns	Medium
BC CITIZ COMM	Baja California State Citizens' Committee, a civil society independent organisation	Pro Protected Area- Protests, political activism, public opinion campaigns	High
FED CONG	Federal Congress of the Union. All the political parties.	Pro Protected Area - Formal requests to protect the islands.	High
CONS ISLAS	Grupo de Ecología y Conservación de Islas, A.C. Mexican NGO.	Pro Protected Area – General coordination, legal defence, media	High
ISLAND CONS	Island Conservation, US NGO	Pro Protected Area - Coordination in the US and conservation research	Medium
TELEVISA	Televisa, a national TV broadcasting company	Pro Protected Area - National news broadcasting at peak hours	High
LOC FISH	Fishermen Cooperatives Regional Federation (FEDECOOP)	Pro Protected Area – Activism	High
BUSI CHAM	Baja California State Business Chamber, Formal Organisation	Against LNG facility	Medium
INTERIOR	Ministry of the Interior, Federal Government	Pro Natural Protected Area – Information	Medium
NAVY	Mexican Navy, Federal Government	Pro Natural Protected Area – Information	Medium

in the Pacific Ocean off Baja California, including the Coronado Archipelago. However, in March 2005 the Communications and Transportation Ministry granted a 30-year renewable lease to ChevronTexaco de México. The LNG plant was immediately viewed by some sectors in Mexico as a potential target for terrorists, a threat to territorial sovereignty, and a threat to the islands' natural resources. The challenges raised by the lease aligned diverse players in complex ways, created novel forces and alliances, started a new social movement and generated intense press coverage in both countries (Lindquist 2004). After years without street protests, there were marches against the proposed plant in the cities of Baja California.

A general sociogram (Moreno 1934; Aguirre-Muñoz 1998; De la Rosa *et al.* 2005) defined the confrontations and linkages of the stakeholders by geography, nationality, attitude towards the LNG facility, and social affiliation, such as civil society, government, and academia (Fig. 3). The sociogram illustrates how the conflict did not follow a simple division between the USA and Mexico. On the contrary, stakeholders on both sides of the border favoured or opposed either the LNG facility or the new protected area (Table 1). Organised fishermen, represented by their Regional Cooperatives Federation (FEDECOOP), actively promoted the new protected area. There was also international activism. Mexican and USA members of Greenpeace, together with other activists, protested at a ChevronTexaco stakeholders meeting in San Francisco. An alliance developed in favour of the protected area and against the LNG facility, encompassing fishermen organisations, conservation NGOs, some federal government agencies, academic institutions, and the local civil society. Important media were sympathetic to the social movement, with national TV coverage at prime time. The lease on Coronado Island was presented at peak hour by the largest TV broadcasting company in Mexico, Televisa, as “a theft from the Nation”.

Legal procedures in Mexico against the LNG lease were ignored by the judicial system, so a request to review the case was sent by US and Mexican citizens to the Commission for Environmental Cooperation (CEC), part of the Free Trade Agreement (NAFTA) between Mexico, the USA and Canada. In 2005, the CEC Secretariat in Montreal, Canada ruled that the Citizens’ Petition fulfilled the required terms and requested a response from the Mexican Government (CEC 2005). While not legally binding, the resolution possesses moral strength. In early 2007 the LNG project suddenly ceased. ChevronTexaco informed the Mexican Environmental Ministry that “ChevronTexaco has decided, because it is convenient to its own interests not to continue with the authorised project …”. Legal protection of the islands has since advanced, with active backing from organised fishermen, the Protected Areas Commission and NGOs. Public hearings concluded and a Conservation and Management Plan draft already exists. The eventual decree has become a public presidential commitment. Networks supporting long-term conservation of the islands saw threats to conservation values and long-term access to fisheries by local fishermen, resulting in a new alignment between conservation NGOs and fishermen. Local communities had an opportunity to understand and appreciate the islands’ wildlife while conservation organisations became more empathetic to the needs and perspectives of local communities.

An organisation to eradicate invasive species from Mexican islands

In addition to the conditions already outlined, one factor has fundamentally affected the success of island restoration through the eradication of invasive pests: a specialised organisation to undertake the complex work. Most eradications of invasive species from Mexico were conducted by GECI, which was formally integrated in 1998. Until 2002, the organisation had a loose structure. By then, full time staff comprised two persons: an improvised manager and a hunter / trapper.

After 2002, a more systematic and strategic organisation was developed. Each employee was hired within a predefined profile based on a specific need or function, and after competing for the job. Key roles and job descriptions followed practical field activities. Currently, the organisation has 24 full time employees with 15 multifunctional biologists and oceanographers as core professional staff. These are supported in the field by seven technicians with skills in animal management, but also able to drive and maintain vehicles, undertake trapping, hunting, and telemetry, and to assist with the aerial dispersion and monitoring of baits. Everyday management is performed by a professional manager and an accountant. Of the professional staff, nine are women, and eight have postgraduate qualifications in biology, ecology, or natural resource management.

There are four main project teams: Guadalupe Island, Marine Birds, Wild Fauna and Rodent Eradications, and Tropical Islands. However, depending on work load, the teams regroup, which enables several projects to run simultaneously (Table 2). A high level of flexibility and skill is promoted by ensuring that the biologists and the technicians know all of the islands where GECI has worked, and through collaborative work on islands in other countries.

Biologists and technicians with ability and experience represent GECI’s most valuable asset. Keeping them and increasing their capacity to restore all of the Mexican islands is a crucial challenge. Additional skills are now being gained within GECI by facilitating postgraduate research on questions derived from applied conservation work. One biologist recently returned to the organisation after completing an MSc degree at the *Instituto de Ecología* investigating food webs on San Pedro Mártir and Farallón de San Ignacio desert islands, where ship rats were recently removed using aerial bait dispersion (Rodríguez Malagón 2009). Two project directors are attending PhD programmes on invasive species on Mexican islands at the University of Auckland (New Zealand) and supported by scholarships from the National Science Council of Mexico (CONACYT).

GECI now has specialised field and office equipment, a biological field station on Guadalupe Island, and a building in Ensenada, Baja California that hosts offices, workshops, vehicles and a warehouse. The total value of the assets has increased from close to zero in 2002 to $US 915, 000 in early 2010.

The organisation is officially authorised by the Mexican federal tax system to receive deductible donations. GECI is registered with the National Science Council, which enables the organisation to bid when proposals are requested by the Council.

Table 2 Invasive species eradication projects and associated activities under way on Mexican islands during the first semester of 2010.

Island	Project	Activity
Socorro, Revillagigedo Archipelago (remote oceanic Pacific tropical island)	Sheep eradication	Ground hunting, last phase
Guadalupe (oceanic Baja California Pacific island)	Comprehensive restoration	Vegetation recovery monitoring - post goat eradication; feral cat control; bird monitoring
Banco Chinchorro (coral cay on the Caribbean)	Feral cat eradication	Full assessment, baseline and eradication preparations
Asunción and San Roque, Baja California Pacific island	Seabird restoration (post feral cat eradication)	Social attraction techniques
San Benito Oeste (Baja California Pacific island)	Introduced mouse eradication	Full assessment, baseline and eradication preparations
San Pedro Mártir, Farallón de San Ignacio, and Isabel (Gulf of California islands)	Ship rat eradications	Post eradication (2007 to 2009) monitoring
Alacranes (Caribbean island)	Ship rat eradication	First assessment
Islas Marías Archipelago (tropical Pacific)	Goat eradication on Maria Cleofas	Baseline and eradication executive plan

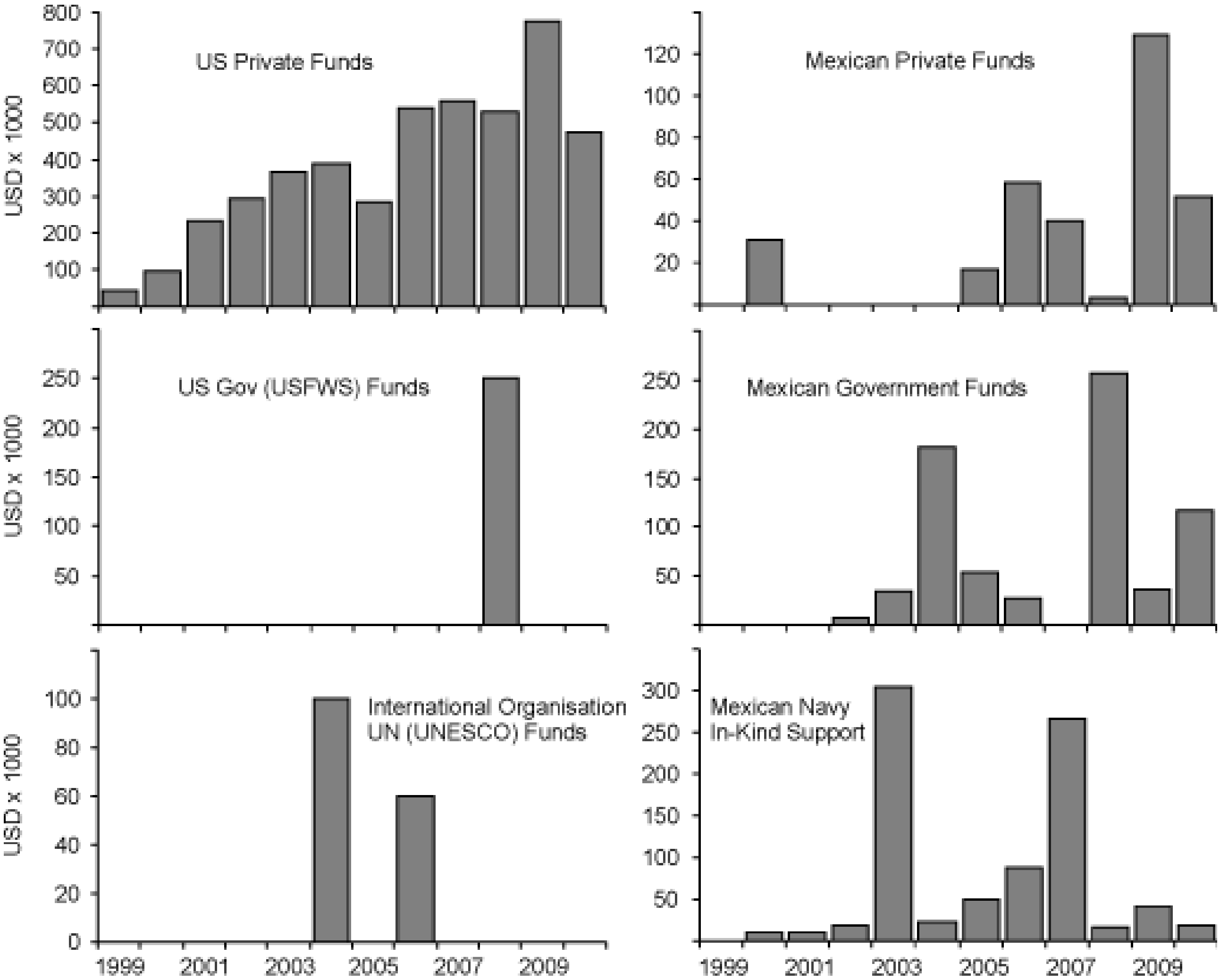

Fig. 4 Time series from 1999 to 2010, showing the origin of funds that enabled the eradication of 48 populations of invasive mammals on 30 Mexican islands.

Finances for eradications of invasive species

Funding obtained by GECI has on average increased since inception, but the sources are variable and the funds are insufficient to enable some of the more challenging projects. Other prerequisites for such projects, including capacity, collaborative networks, government support and permitting, and available techniques, are in place or are readily accessible. Insufficient resources to retain key personnel would severely threaten continued restoration work. Alternatively, sufficient and sustained funding could enable an unprecedented opportunity for the restoration of all the Mexican islands by 2025, a globally important strategic goal and a viable achievement.

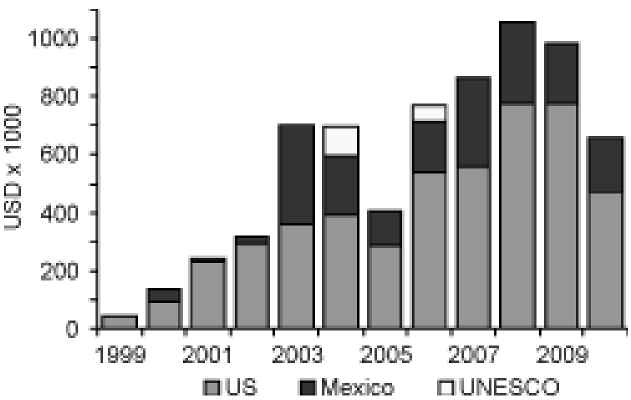

Fig. 5 US, Mexico, and UNESCO funding to eradicate invasive species on Mexican islands.

Between 1999 and 2010, approximately $US 7 million has been invested in eradications on Mexican islands, with funds from Mexico, the US, an international organisation (UNESCO) and in-kind support from the Mexican Navy. The figures also include work on pre-eradication assessments, eradication planning and post-eradication monitoring (Fig. 4).

By country of origin, 70.1% of the total cumulative resources was provided by US donors, largely from private foundations (Fig. 5). Mexico contributed 27.6%, half of which was 'in-kind' contributions from the Mexican Navy through support given by large vessels during eradications, regular transportation to islands, logistic support and use of their infrastructure facilities.

Mexican federal government agencies within the sphere of the Environment and Natural Resources Ministry (SEMARNAT) include the Biodiversity Commission (CONABIO), Natural Protected Areas Commission (CONANP) and National Institute of Ecology (INE). Collectively, these have contributed 10.4% of the total invested in eradication projects on islands during the last decade.

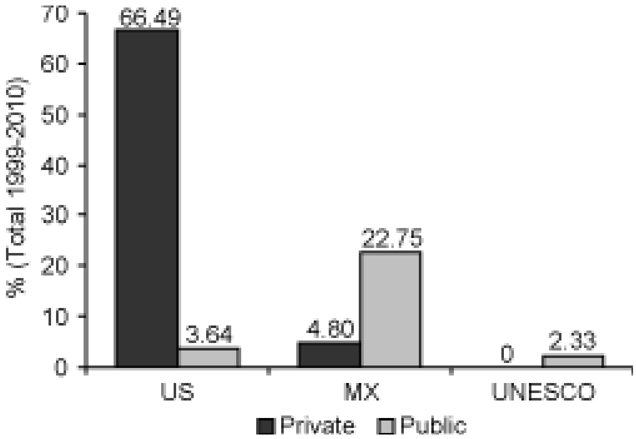

Fig. 6 Private and public contributions to eradicate invasive species on Mexican islands from 1999-2010.

Fig. 7 The Mexican Navy MV Sonora supporting the ship rat eradication on San Pedro Mártir Island, Gulf of California, October 2007. The operation used Cl 25 (Bell Labs) brodifacoum bait spread by a helicopter from Aspen Helicopters, USA, and a special bucket from Helicopters Otago, New Zealand.

So far, the private Mexican sector has contributed 4.8% of the total. Support from international organisations has been 2.3% of the total provided by UNESCO for a goat eradication on the Espíritu Santo Island. The straightforward eradication, was stopped when almost completed following political challenges during the 2006 presidential election.

A 'one-off' US $500,000 US-Mexico bi-national fund was established in 2008 to eradicate invasive species on Mexican islands as a means of protecting migratory species of common interest, mainly migratory birds. Half of the resources were granted by the US Fish and Wildlife Service and the other half by CONANP. The fund was operated during 2009 by GECI under supervision of CONABIO. Four projects were successfully undertaken (Table 2): the eradication of sheep (*Ovis aries*) from Socorro Island; the eradication of ship rats from Isabel Island; an eradication plan for feral cats (*Felis catus*) from Guadalupe Island; and a workshop on island invasives for federal government staff, including CONANP, INE, CONABIO, the Mexican Navy, SEMARNAT and fishermen co-operatives.

Although funding has followed a positive growth trend during the last decade, the funds available during 2009 and 2010 are lower than the immediate previous year, reflecting reduced private donations from the US. These donations are still the largest component of private funding to date (Fig. 6).

Beginning in 2009, a combination of federal and private funds became available to conduct eradications on islands through the Mexican Fund for the Conservation of Nature (FMCN) as part of the National Fund for Protected Areas (FANP). This fund is being maintained and expanded, providing now opportunities for multi-annual support. If the pace is sustained and funding is assured a strategic goal could be met: to eradicate all invasive mammals from the remaining Mexican islands by the year 2025.

Institutional support for eradications of invasive species

The positive results from eradications on Mexican islands have caught the attention of several state and federal government agencies. The Congress of the Union has been involved with invasive species. As a result, the following policy instruments and partnerships provide a framework for the eradication of invasive species.

Firstly, CONABIO recently completed a National Strategy on Invasive Species: Prevention, Control and Eradication (CONABIO 2010). Public hearings have concluded and an Advisory Committee is in place to implement the strategy. CONABIO has also co-ordinated a State of the Nation analysis (Sarukhán Kermes 2009), where a comprehensive chapter on invasive species identifies islands as a special case (Aguirre-Muñoz and Mendoza Alfaro 2009).

Secondly, in 2008 CONANP, which is part of the Environment Ministry (SEMARNAT), formed a compact department that deals with introduced species with emphasis on island eradications. Departmental personnel assist by pursuing permits from the Wildlife General Directorate and facilitating inter-institutional collaboration. In March 2010, CONANP approved Guidelines to Prevent, Control and Eradicate Invasive Species on Insular Federal Natural Protected Areas (CONANP 2010), thereby officially supporting eradications projects on islands.

Thirdly, the Ministry of Interior (SEGOB) has a special office – Subdirección de Administración del Territorio Insular – to deal with general governance issues in Mexican federal insular territory. With a constitutional mandate to manage the Mexican islands, this office facilitates relationships with the Mexican Navy, and has the legislative power to provide any general permits required in support of those granted by the Environment and the Health Ministries.

The Mexican Navy provides essential support for eradication activities on a case-by-case basis. For example, a helicopter was deployed on Guadalupe Island by a Navy vessel with a platform and hangar. Hunters were transported repeatedly to Socorro Island to remove sheep, while ammunition and conservation personnel were transported between the mainland and Socorro Island by helicopter; a 3.5 hour flight. The MV Sonora and its crew supported the helicopter-based eradication of ship rats on islands from the Gulf of California (Fig. 7). Navy lodging facilities on the islands are offered to the scientists and technicians that do the eradication work. In these examples, the Mexican Navy takes care of natural capital in a novel and productive way of attending to sovereignty.

The National Institute of Ecology (INE), following a research perspective, has been a long time partner in eradication projects. Their first financial investment was for goat eradication from Guadalupe Island. Over the last three years, INE has coordinated a project with the Public Security Ministry and GECI to assess the invasive species situation on the Islas Marías Archipelago, in preparation for systematic eradication of invasive vertebrates. INE, with other government agencies and GECI, is also starting the integration of a "National Strategy for the Conservation and Sustainable Use for the Mexican Islands".

Overall, the government's approach to introduced species has shifted from regulatory, to proactive facilitation. Institutional development, the creation of federal protected areas, and the generation of new policy instruments, indicate that eradications, particularly on islands, have gained widespread institutional support.

CONCLUSIONS

The historical perspectives outlined here follow remarkable changes in values, attitudes, discourses and practices towards islands by the 'social actors' in Mexico, with particularly rapid change over the last three decades. Beginning with abuse of island resources, abandonment of remote territories, and then questionable dealings by the state over aspects of sovereignty, attitudes have since been transformed. These changed attitudes were illustrated recently when a proposed LNG plant near one island became linked to perceived threats to sovereignty and stimulated a national conservation movement.

Changing attitudes are sometimes influenced by chapters of history linked to national or global events. In Mexico, these events included fragility of the new nation during the 19th and early 20th centuries, the international adoption of the Exclusive Economic Zone (EEZ), and recent pressures and opportunities presented by globalisation. In addition to events that shaped attitudes within Mexico, other components of change echo those found elsewhere. The way that recent attitudes towards island conservation were transformed into a sustained '*Leitmotiv*' by a non-governmental environmental organisation reflects gaps and delays between the concerns of a dynamic civil society and the corresponding more rigid government agencies and agendas (Giddens 1998). Nonetheless, issues that are of enduring concern to civil society are eventually incorporated into government agendas. The process this followed for island conservation in Mexico began with raised awareness of the issues, continued by social acceptance, scientific research, organisational development to exploit identified needs, securing of funding to support projects, and finally, institutional acceptance and support.

Awareness is a particularly important component, and stemmed from recognition by civil organisations and academic institutions of the great ecological value and fragility of Mexican island ecosystems and biodiversity, sustainable practices that could be followed by local communities such as fishermen co-operatives, and the need for government agencies to pro-actively strengthen sovereignty and make good use of national territory.

Recent and efficient eradications of invasive mammals on Mexican islands have been central to a new, caring attitude towards the Mexican insular territory. This new attitude developed as a 'bottom-up' social construction that then spread to a complex suite of diverse social actors. Central to this success has been the development of an NGO, GECI. Although focused on the eradication of invasive species on islands, this organisation has built collaborations with government agencies and local communities such as fishing villages.

This crucial role of NGOs in conservation of natural resources in Mexico has parallels in other federal government systems such as the USA, where Wilson (2002) regarded them as the spearhead of conservation movements. As in Mexico, the most successful model identified by Wilson (2002) involved strong relationships between the private sector, government and science and technology. In Mexico, protection of biodiversity, attention to sovereignty and good use of natural resources formed a simple philosophical triad that produced outstanding results for island restoration and conservation. There is little need to change this approach, but it could be reinforced.

As Wilson (2002) also recognised, finances are a crucial issue for NGOs. Secure funds and retention of experienced personnel are prerequisites if we are to meet our goal of restoring the remaining Mexican islands still inhabited by invasive mammals. Funds coming from outside Mexico through private US donors will need to be maintained and increased while funds from within Mexico are developed. Mexican public funds for island restoration should also grow consistently and significantly. An investment of approximately two million dollars per year over the next 15 years is needed to eradicate the remaining invasive vertebrates on Mexican islands.

The effects of these restorations should not be limited to positive outcomes for biodiversity. They can also provide an incentive to use models for sustainable development. Compared with the mainland, Mexican islands are well suited to such an approach. The islands are self-contained, the actors are few, governance is high, social aspects are simpler and ecosystems are also less complex than on the mainland. Abalone and lobster poachers do not make it to the islands or are relatively well controlled. Green certificates such as those granted by the Marine Stewardship Council can be achieved for all of the island fisheries. This movement to sustainable use has already started, adding value to products in the markets and increasing consciousness of local fishermen communities. Careful use of the natural resources on islands can then become an element of pride and territorial identity. The possibility of switching fully to alternative energies such as solar and wind is a viable option, as most of the communities are small and industrial needs are few. Increased understanding through quality education about biology, ecology and sustainable development can be offered and developed on islands as in few other locations. Restoration and management models can be researched, understood and applied on islands where there are fewer variables than on the mainland but with prospects of relatively effective control over them.

A successful, well documented and well understood story around restoration and sustainable development on islands could inspire similar work on larger scales and on continental territories. Few places in the world are at present improving all aspects of their natural, social and financial capital. Mexican islands are.

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Cat impact and management on two Mediterranean sister islands: "the French conservation touch"

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Abstract Feral cats (*Felis catus*) are one of the most damaging introduced species for island species worldwide. While cat control or eradication is handled with increasing efficiency on uninhabited islands, the strong bond with humans, regardless of ownership, makes cat management difficult on inhabited islands. We conducted a cat-removal programme on Port-Cros Island where both the presence of humans and their cats threaten *Puffinus yelkouan*, an endangered Mediterranean endemic species of burrowing petrel. The two largest French-breeding colonies of this procellariid are on the two studied islands: Port-Cros and Le Levant. The cat-removal programme was implemented on Port-Cros, with Le Levant used for comparison. Cat diet studied through scat analysis showed cats to be responsible for killing 162 ± 46 and 21 ± 4 shearwaters per cat and per year on Le Levant and Port-Cros respectively. Bird breeding parameters were monitored during seven years on Port-Cros (before and after cat removal) and three years on Le Levant. By constructing a shearwater population viability model, we calculated that the cat impact on the yelkouan shearwaters threatens the entire population in the long term and justified cat removal. We designed a conservation management plan for Port-Cros where, taking into account human presence, feral cats were live-trapped and domestic cats were sterilised. Following this two year campaign, cat predation of shearwaters ceased, followed by an increase in the shearwater breeding population. Thus, protecting seabirds from cat predation is possible, even on islands where inhabitants are notoriously reticent to any sort of cat removal programme.

Keywords: Feral cat, *Felis catus*, eradication, yelkouan shearwater, *Puffinus yelkouan*, island conservation

INTRODUCTION

The spread of non-indigenous species is considered second only to habitat destruction in harming native communities and considered first to impact island biodiversity (Vitousek *et al.*1995; Williamson 1996; Whittaker and Fernández-Palacios 2007).

Cats (*Felis catus*) were first introduced to islands in the Mediterranean in 9000 BP (Vigne *et al.* 2004; Driscoll *et al.* 2007), and have since been introduced to islands worldwide from the sub Antarctic to the sub Arctic, including the most arid and mesic islands (Ebenhard 1988; Courchamp *et al.* 2003). They are successful invaders of islands because they can survive without access to fresh water, have high fecundity, a high adaptability to novel environments, and have generalist predatory behaviours that allow them to feed on most prey species (Pearre and Maass 1998; Fitzgerald and Turner 2000; Say *et al.* 2002). Cats are one of the most damaging invasive predators on islands (Fitzgerald 1988; Macdonald and Thom 2001) and are responsible, at least in part, for 8% of global bird, mammal and reptile extinctions and a significant threat to almost 10% of critically endangered birds, mammals and reptiles (Medina *et al.* 2011).

Seabirds are often badly affected by cat introduction on islands (Courchamp *et al.* 2003; Blackburn *et al.* 2004; Donlan and Wilcox 2008), particularly petrels and shearwaters, due to their lack of predatory defence and their high vulnerability to adult mortality (Brooke 2004; Le Corre 2008). Different studies have recently shown that several *Puffinus* species, especially those belonging to the Manx shearwater *P. puffinus* worldwide 'complex', are seriously threatened by introduced predators (Mayol-Serra *et al.* 2000; Ainley *et al.* 2001; Cuthbert 2002; Keitt *et al.* 2002; Martínez-Gómez and Jacobsen 2004). The Yelkouan shearwater (*Puffinus yelkouan*) is endemic to the Mediterranean Basin and near threatened and declining (IUCN Red List), with a breeding population possibly not exceeding some thousands of pairs and probably restricted to a few breeding locations, most of which have introduced predators (Bourgeois and Vidal 2008).

Eradicating cats from islands can protect native species from the threat of extinction (Nogales *et al.* 2004) and research on the ecology of insular feral cats can improve the efficacy and prioritization of cat eradications (Fitzgerald 1988; Paltridge *et al.* 1997; Fitzgerald and Turner 2000; Macdonald and Thom 2001).

The Hyères Archipelago has domestic and feral cat populations, and is a major breeding site for Yelkouan shearwater. We studied shearwater population viability in order to conduct relevant feral cat management.

The aims of this study were to: 1) monitor the shearwater populations; 2) study cat diet in relation to the shearwater breeding cycle; 3) evaluate the cat impact on the population viability of shearwaters; and 4) manage cat populations in order to maintain biodiversity on islands.

MATERIALS AND METHODS

Study area

This study was conducted on two islands within the Hyères Archipelago located in the north-western Mediterranean Sea (Fig. 1). Le Levant Island (10.8 km²) has a maximum elevation of 140 m above sea level and is 9.15 km from the mainland. It is a military island for 90% of its area; the remaining 10% is occupied by civilians. Port-Cros Island (6.40 km²) has been protected by National Park status since 1963, has a maximum elevation of 196 m above sea level, and is 15 km from the mainland. The climate is sub-humid, temperate Mediterranean with an average annual rainfall of 582.4 mm and an average annual temperature of 16.5°C (Levant Island Meteorological Office, 1997–2007). The islands are siliceous, Le Levant being mainly covered by the typical shrubs of "maquis" vegetation with sparse sclerophyllous oaks (*Quercus ilex*) and halepo pines (*Pinus halepensis*); Port-Cros being covered by mixed forests of the sclerophyllous oaks and halepo pines.

These islands have long been home to introduced vertebrates including cats for two centuries (Pasqualini 1995), rats (*Rattus rattus*) at least since the Roman period (Ruffino and Vidal 2010), and rabbits (*Oryctolagus cuniculus*). The Mediterranean endemic seabird, yelkouan shearwater is represented on Le Levant by 800-1,300 pairs and on Port-Cros by 140-180 pairs from a world population likely to be fewer than 15,000 pairs (Bourgeois and Vidal 2008).

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Shearwater monitoring

We monitored 100 shearwater burrows on Port-Cros during seven breeding seasons (2003 to 2009) and in 76 burrows during three breeding seasons (2007 to 2009) on Le Levant to record the percentage of occupied burrows and breeding success. Like most seabirds, yelkouan shearwaters have low reproductive output; they start breeding at around 6 years of age, generally first attempts to breed fail, and they produce only one egg per year (e.g., Brooke 1990). They arrive at their breeding sites in late October or early November (Vidal 1985; Zotier 1997), which corresponds to the prospecting period when birds visit the burrows and look for their mate. Egg laying is from mid-March to early April, hatching in May and fledging in July and early August.

A miniature infrared camera on a stiff coaxial cable was “snaked” down each burrow to determine the presence of pairs, eggs or chicks (Bourgeois and Vidal 2007). Burrows were checked nine times during each breeding season: at the end of the pre-laying period, the start, middle and end of the laying and hatching periods, and 15 days before the beginning and at the middle of the fledging period. A last check was done at the end of the breeding season to find possible corpses and confirm chick fledging (Bourgeois 2006). A randomisation test was used to compare the percent of occupied cavities between the first year and the last year of our censuses

Cat diet study

The diet of feral cats was studied through scat analysis (Fitzgerald *et al.* 1991; Bonnaud *et al.* 2007). We opportunistically collected scats on sample paths from October 2002 to August 2004 on Port-Cros and from October 2006 to August 2008 on Le Levant. Scats were collected five times per year: when the shearwaters were prospecting, breeding, hatching, rearing and during their annual exodus. By removing all scats found in the field and excluding very old ones, we assumed that each sampling set represented the cat diet for that period. All scats found were reported on a map with a handheld global positioning system. This sampling allowed us to determine the cat diet during each of the shearwater breeding phases.

Scats were analysed by washing through a 0.5-mm sieve under a stream of hot water and separating all items such as hairs, feathers, bone fragments, teeth, and insect chitin (Nogales *et al.* 1988). Each item was then identified by comparison with reference material. The diet results were given in frequencies of occurrences and numbers of prey. A Pearson χ^2 test for independent samples was used to test the difference of the cat diet on both islands, then randomisation tests were performed to detect differences in cat consumption of each prey thereby allowing comparison of small percentages (PD = observed percentage differences; Manly 1997).

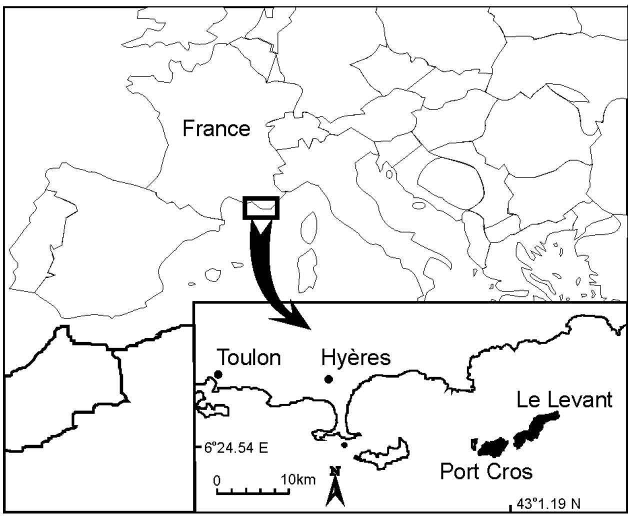

Fig.1 Study site, Hyères archipelago (south east of France). Study conducted on Port-Cros and Le Levant Island.

Cat impact on yelkouan shearwaters

To estimate the magnitude of cat predation on shearwaters, we first calculated the number of shearwaters eaten each year by the cat population. Since no identical parts from two or more shearwaters were found in any one cat scat, each scat were assumed to be of one bird (Keitt *et al.* 2002; Cuthbert 2002; Bonnaud *et al.* 2007). Cats usually defecate once per day (Konecny 1987). Thus, the mean number of the shearwaters per scat is equivalent to the mean number of shearwaters ingested per day and per cat ($NP_{/d}$). The annual mean number of shearwaters killed on Le Levant (NP) by the cat population was calculated as follows:

$NP = NP_{/d} \times 365 \times N_{cat}$ (1)

with N_{cat}: number of cats on the island.

Predation rates were calculated assuming: 1) predation on prospectors (birds looking for a mate and a burrow) (PB) was four times higher than on breeders (birds which were breeders the next year and the current year) (PP); and 2) predation was exerted on prospectors from age 3 (from $N3_P$ to $N6_{+P}$) and on breeders (first breeding assumed at 6 years, (Brook 1990)) ($N6_{+B}$)

$NP = P_B \times (N6_{+B}) + P_P \times (N3_P + N4_P + N5_P + N6_{+P})$ (2)

With $P_P = 4 \times P_B$

And NP = number of shearwaters killed per year.

The impact of cat predation on shearwater population dynamics was assessed by constructing a shearwater demographic population model adapted from Bonnaud *et al.* 2009 (see Appendix for the model structure and implemented parameters). The value of shearwater breeding success without cat predation came from monitoring burrows in Port-Cros colonies during four breeding seasons (2006 to 2009). Cat predation rates were then included in several scenarios depending on cat population estimates and taking into account the higher shearwater population estimates for the both islands (shearwater population of 1) Le Levant Island = 2600 breeders and 2) Port-Cros Island = 360 breeders; Bourgeois and Vidal 2008). The demographic population model was run with ULM (Unified Life Models) mathematical modelling software (Legendre and Clobert 1995) and we conducted Monte Carlo simulations (100 time steps and 1000 trajectories) to account for the uncertainty of several population parameters.

Cat management on Port-Cros Island

Cat presence on Port Cros constituted a threat to the shearwater population which, at 180 pairs, was already small. A cat management programme was started in January 2004. The presence of human inhabitants, and domestic cats meant that the removal of feral cats should be undertaken using only non-lethal methods, i.e. cat living-traps checked each morning and evening. Complete cat eradication was not possible due to the persistence of a small domestic cat population located in the village. The trapping campaign was initially concentrated near the shearwater colonies and then extended along all paths, especially where cat scats were found. A sterilisation campaign was conducted on the domestic cats and all new domestic cats arriving were checked for sterilisation. During and after cat control we collected cat scats during selected phases of the shearwater breeding cycle. We used a Mann-Whitney U test to compare the number of scats found before the beginning of the cat control and after the last feral cat was caught. These scats were analysed only in order to detect shearwater remains.

Table 1 Monitoring of the breeding parameters of the yelkouan shearwater and the percent of occupied nests on Port-Cros and Le Levant Islands.

	Port-Cros Island (360 breeding birds, 100 burrows monitored)							
Year survey	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**mean ± SD**
Occupied burrows	28	32	41	42	39	40	37	37
% occupied burrows	27.7	31.1	39.8	39.6	37.5	38.8	36.6	35.9 ± 4.7
Hatching success*	70.0	85.7	97.4	89.5	73.7	94.7	91.7	86.1 ± 10.5
Fledging success*	92.9	95.8	83.8	85.3	92.9	91.7	100.0	91.8 ± 5.7
Breeding success*	65.0	82.1	81.6	76.3	68.4	86.8	91.7	78.9 ± 9.6
	Le Levant Island (2600 breeding birds, 76 burrows monitored)							
Year survey					**2007**	**2008**	**2009**	**mean ± SD**
Occupied burrows					33	32	30	32
% occupied burrows					46.5	42.1	41.7	43.4 ± 2.7
Hatching success*					93.8	87.1	93.3	91.4 ± 3.7
Fledging success*					76.7	88.9	89.3	84.9 ± 7.2
Breeding success*					71.9	77.4	83.3	77.5 ± 5.7

* Shown as a percentage of the occupied burrows.

Due to the renowned harmful effect of rats on seabirds (e.g., Jones *et al.*, 2008) we tested whether cat control affected rodent numbers. As cats preyed mainly upon rats on this island (Bonnaud *et al.* 2007) a meso-predator release was possible. We set two lines of 30 traps in two different areas of the island and set live traps every 10 meters during four consecutive nights for 19 trapping sessions from December 2004 to August 2008 at three or four months intervals

On Le Levant, other than an awareness campaign about the threat of feral cat presence for island biodiversity, there has been no cat management.

RESULTS

Shearwater monitoring

Shearwater breeding, monitored on Port-Cros from 2003 to 2009 and on Le Levant from 2007 to 2009, showed high breeding parameter values (Table 1). On both islands the percent of occupied nests was low (36% on Port-Cros and 43% on Le Levant). During the study period, nest occupation significantly increased on Port-Cros (PD = -0.149, p = 0.0130) and decreased, but not significantly, on Le Levant. Hatching success increased on Port-Cros and remained stable on Le Levant. Fledging success was high on both islands but slightly higher on Port-Cros where success in the last year sampled reached 100. The overall breeding success increased to reach similar values on both islands.

Cat diet study

We collected and analysed 689 scats on Port Cros and 200 on Le Levant. Cats on both islands preyed mainly upon introduced mammals. Yelkouan shearwater was the most frequent bird found in the scats (Table 2). Other birds (mainly passerines), reptiles and invertebrates were secondary prey. When all prey consumed was considered, significant differences appeared between the cat diets of both islands ($\chi^2 = 314$, $p < 0.001$). The consumption of rabbits (PD = 0.203, $p < 0.001$) and shearwaters (PD = 0.376, $p < 0.001$) were significantly higher on Le Levant than on Port-Cros and consumption of rats (PD = 0.350, $p < 0.001$) and wood mice (*Apodemus sylvaticus*) (PD = 0.322, $p < 0.001$) were significantly lower. More than one mammal per scat was found in scats from Port-Cros, mainly rats and wood mice. Less than one mammal per scat was found in scats from Le Levant, the cat diet being mainly comprised of rabbits and shearwaters.

Regarding cat predation on shearwaters, frequency of occurrence was low on Port-Cros Island (shearwater remains appeared in 5.9% of scats found) compare to that on Le Levant Island (shearwater remains appeared in 44.3% of scats found).

Cat impact on yelkouan shearwaters

The number of shearwaters eaten per cat per year reached 162 ± 46 and 22 ± 4 individuals respectively on Le Levant and Port-Cros. Peaks in predation on shearwaters on both islands were during autumn and winter (October–November and December–February), corresponding to their prospecting period (Fig. 2) and this predation remained high during spring on Le Levant (Fig. 2B).

Table 2 Food categories of the cat diet on Port-Cros and Le Levant Islands expressed as frequency of occurrence and the numbers of prey per scat.

	Port-Cros Island (August 2002 - August 2004)		Le Levant Island (August 2006 - August 2008)	
Food categories	**Frequency of occurrence (%)**	**Number of prey per scat***	**Frequency of occurrence (%)**	**Number of prey per scat**
MAMMALS	91.87	1.57	74.50	0.75
Rattus rattus	77.94	0.95	43.00	0.45
Apodemus sylvaticus	34.69	0.54	2.50	0.03
Oryctolagus cuniculus	6.68	0.09	27.00	0.27
BIRDS	16.69	0.12	51.00	0.51
Puffinus yelkouan	5.81	0.05	43.50	0.44
other-birds	10.89	0.06	7.50	0.08
REPTILES	7.84	0.03	11.50	0.12
INSECTS	11.03	0.05	8.50	0.11

* data only available between August 2003 and August 2004

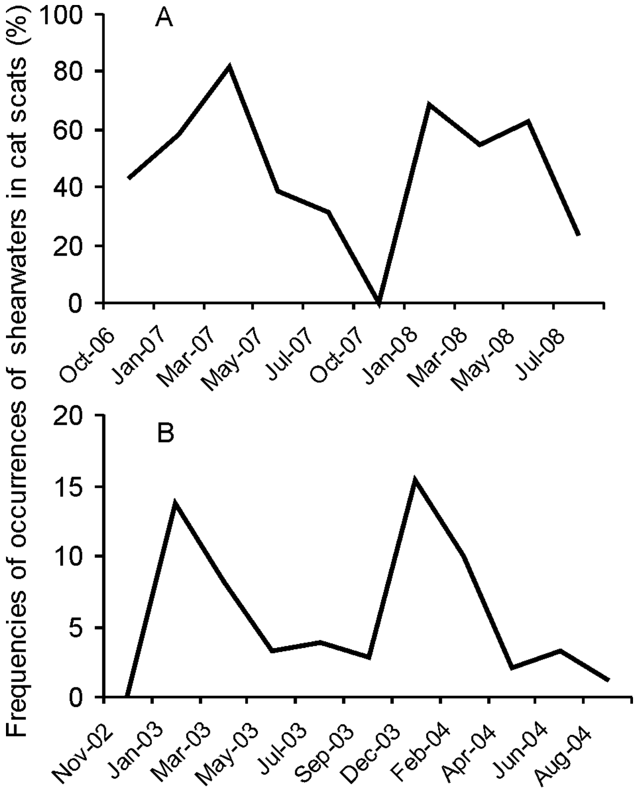

Fig. 2 Frequencies of occurrences of shearwater remains found in cat scats during a 2-year survey on (A) Le Levant Island (B) on Port-Cros Island.

The population of cats on Port-Cros was estimated as 20 based on trapping data during feral cat removal (Bonnaud *et al.* 2010). It was impossible to estimate the cat population of Le Levant but the small number of scats found per sampling period suggested that cat density on this island was lower than on Port-Cros. Thus, we tested three scenarios of 5, 10 and 20 individuals (Table 3). Applying equations (1) and (2) we calculated the number of shearwaters killed per year by cat populations of both islands and the predation rates on breeders and prospectors (Table 3).

The shearwater demographic population models were run using scenarios predicting that: 1) without cat predation the shearwater populations of both islands showed growth rates higher than 1, and 2) with cat predation all scenarios showed decline leading to eventual extinction of the shearwater population.

Cat management on Port-Cros Island

Cat removal started in January 2004, with 28 cats trapped over two years (Table 4). Trapping success progressively

Table 4 Numbers of trap nights and cats trapped during the cat management program conducted on Port-Cros Island.

Period	Trap nights	Cats caught
Dec-Feb 04	45	2
Feb-Apr 04	41	4
Apr-Jun 04	89	2
Jun-Aug 04	60	1
Aug-Oct 04	66	3
Oct-Jan 05	190	8
Jan-Mar 05	262	4
Mar-May 05	134	1
May-Aug 05	118	1
Aug-Oct 05	132	0
Oct-Jan 06	617	2
Jan-Mar 06	77	0

decreased, becoming nil by January 2006 despite regular trapping sessions being continued. Subsequently, only neutered domestic cats were seen wandering outside the village and were photographed by cameras placed near paths. No sign of recovery of the cat population was observed. The number of scats found on sampling paths significantly decreased from 0.631 ± 0.119 scats/day before the beginning of cat control to 0.177 ± 0.022 scats/day after the last feral cat was caught ($U = 2$, $p < 0.001$). Between August 2004 and August 2005 only one scat was found (in May) and it contained shearwater remains. Cat scats found after August 2005 were assumed to belong to the few domestic cats wandering around the island but without evidence that they are preying upon shearwaters

Rat trapping success in trap lines varied between seasons and years but remained low during both 1978-1987 (mean: 0.068 ± 0.024 rats caught per trap-night , Granjon and Cheylan 1993) and 2004-2008 (mean: 0.112 ± 0.026 rats caught per trap-night, this study) monitoring periods.

DISCUSSION

Shearwater monitoring

Yelkouan shearwater breeding populations were reduced to a few individuals, especially on Port-Cros, due to predation by cats. Bourgeois and Vidal (2007) and Bourgeois *et al.* (2008b) showed that these breeding habitats are far from saturation. Both have unoccupied burrows within colonies and sites suitable for new colony establishment. Cat predation kills more shearwaters when they are in the prospecting stage of the breeding cycle. As breeders they spend little time on the ground and avoid predation by rapidly entering their burrows (Bourgois *et al.* 2008a). Despite the presence of predators the breeding populations of shearwaters on both islands show high

Table 3 Results of the shearwater demographic models which include cat predation rates according to the size of the cat population on Port-Cros and Le Levant Islands. $Shearwater_{Pop}$: size of the shearwater populations, $N_{shear.killed}$: number of yelkouan shearwater killed per the cat population and per year, Cat_{Pop}: size of the cat populations, PB: cat predation rate on breeding birds, PP: cat predation rate on prospecting birds, λ: growth rate of yelkouan shearwater populations, T_{ext}: predicting time (in years) for yelkouan shearwater population extinction.

	Port-Cros		Le Levant			
$Shearwater_{Pop}$	360		2600			
Cat_{Pop}	0	20	0	5	10	20
$N_{shear.\ killed}$	0	431 ± 72	0	810 ± 230	1621 ± 460	3241 ± 920
PB	0	0.386 ± 0.065	0	0.101 ± 0.029	0.202 ± 0.057	0.403 ± 0.115
PP	0	1.544 ± 0.260	0	0.404 ± 0.116	0.808 ± 0.228	1.612 ± 0.460
λ	1.0102 ± 0.0000	0.7054 ± 0.0064	1.0101 ± 0.0000	0.8586 ± 0.0001	0.6805 ± 0.0021	0.7331 ± 0.0058
T_{ext} (year)	-	6.3780 ± 0.0185	-	53.6820 ± 0.0649	21.1840 ± 0.0671	6.5830 ± 0.0384

reproductive success (77 to 79% Table 1) when compared to other shearwater and petrel populations (Brooke 1990; Hunter *et al.* 2000; Cuthbert 2002; Dunlop *et al.* 2002; Le Corre *et al.* 2002; Jouventin *et al.* 2003; Igual *et al.* 2007; Rayner *et al.* 2007; Pascal *et al.* 2008). Now that predation is controlled, the settlement of new breeders should increase on Port-Cros.

Cat diet study

Our study supported the common observation that feral cats are highly generalist predators, able to feed on prey ranging from small insects to birds and mammals that weigh more than 500 g (Nogales and Medina 1996; Tidemann *et al.* 1994; Turner and Bateson 2000). However, cats can specialise on what is available and only a few species represented the major part of its diet. Introduced mammals and shearwaters were the prey mainly eaten by cats on these Mediterranean Islands. The differences in cat diet between the two islands are explained by the high frequency of occurrences of rabbits and shearwaters on Le Levant and the high frequencies of occurrences of rats and wood mice on Port-Cros (Bourgeois and Vidal, 2008; Port-Cros National Park pers. comm.). Because rabbits and shearwaters are large prey items, the consumption of one constitutes the required daily food intake per cat (Bonnaud *et al.* 2007). In contrast the consumption of rodents (rats and wood mice) generally requires the cat to prey upon more than one individual and can result in greater diversify in the diet. This indicates that the number of prey items eaten may provide a trophic index which can be used to evaluate cat impact on prey population dynamics.

Cat impact on yelkouan shearwaters

The cat diet studies revealed high cat predation during the prospecting period of the shearwaters and continuing predation throughout the year. Cat predation on the shearwaters reached 162 ± 46 and 22 ± 4 individuals per cat per year respectively on Le Levant and Port-Cros, placing these populations of shearwaters at high risk of local extirpation. These islands have the largest colonies of yelkouan shearwaters in France, being one of the largest in the world (Bourgeois and Vidal 2008).

Mathematical population dynamic models are a useful tool to evaluate the impact of species interactions. Our model predicted annual population growth rates slightly greater than one without cat predation, which was consistent with predictions for populations of other *Puffinus* species: *P. griseus* (1.017, Hamilton and Moller 1995; 1.044, Jones 2002), *P. huttoni* (0.930–1.050, Cuthbert and Davis 2002), *P. opisthomelas* (1.006, Keitt *et al.* 2002), *P. auricularis* (1.001, Martinez- Gómez and Jacobsen 2004) and *P. mauretanicus* (1.007, Oro *et al.* 2004). This suggests that the scenario selected for the yelkouan shearwater can be considered realistic and the model structure suitable. Few studies have taken predation on prospecting birds into account. Prospecting birds are probably more vulnerable to cat predation due to their behaviour: wandering on the ground and calling outside burrows, rather than entering the burrow rapidly after landing (James 1985; Brooke 1990; Ristow 1998; Bourgeois *et al.* 2008a; Bonnaud *et al.* 2009). Even with a small cat population included, the shearwater demographic showed a decrease of the shearwater populations. In some cases, cat predation on prospectors was so high it exceeded the number of prospectors available, indicating immigration from outside these populations. In summary, our results showed that: 1) these shearwater populations cannot survive if they are not supported by immigration; and 2) even if the breeding populations have a high breeding success, these small populations seem to be at a high risk of local extinction due to feral cat predation.

Cat management on Port-Cros Island

Faced by the strong threat exerted by cats on the yelkouan shearwaters, a cat management campaign was conducted on Port-Cros Island. This cat management campaign was, to the best of our knowledge, one of the first conducted in the Mediterranean Basin (Genovesi 2005; Lorvelec and Pascal 2005). It was also one of the few successfully developed using only non-lethal trapping and conserving a domestic population of neutered domestic cats on the island (Nogales *et al.* 2004). Non-lethal trapping proved to be successful in eradicating the feral cat population and rapidly prevented cat predation on native threatened species. No feral cats were observed or trapped on the island during nearly three years following the last feral cat caught in October 2005, despite a reduced but continuous trapping campaign. Feral cat control, which started in 2004, resulted in an increase in numbers of occupied shearwater burrows and breeding pairs, confirming that cat predation, being mainly focused on the prospecting period, probably limits the recruitment of young breeders (Keitt *et al.* 2002; Massaro and Blair 2003; Peck *et al.* 2008). Moreover, due to the high probability of a top-down-regulated ecosystem on Port-Cros, the rat population on this island was carefully monitored during and after cat control (Russell *et al.* 2009). Rat-trapping success values have remained similar to previous values recorded before cat control (Granjon and Cheylan 1993). This suggests that cat control, while diminishing predation pressure on rats, has not led to a significant increase in the rat population size, nor their impact on seabirds.

Implications for conservation

On islands with multiple introduced predators and native prey species, it is commonly suggested that the best solution is the simultaneous eradication of both introduced top- and mesopredators to avoid any risk of mesopredator release effect (Simberloff 2001; Zavaleta *et al.* 2001; Courchamp *et al.* 2003; Blackburn 2008). However, when introduced predators threaten long-lived seabirds, top-predators like cats have larger detrimental effects on their population dynamics than mesopredators (Le Corre 2008; Russell *et al.* 2009). Moreover, top-predator populations are not the only means of regulating mesopredator populations (Blackwell *et al.* 2003). Thus, the eradication of top-predators should be encouraged simultaneously with monitoring the population dynamic of other species that can react to this ecosystem management. Knowing that most of the islands of the Mediterranean basin house feral and domestic cats and native endemic species, this study indicates that even if a complete cat eradication is not feasible, feral cat eradication coupled with the persistence of a neutered domestic cat population can lead to the same results as total eradication (Oppel *et al.* 2011). As intraguild predation involves complex mechanisms and often multiple trophic interactions (top-down or bottom up processes) (Fukami *et al.* 2006; Elmhagen and Rushton 2007; Ritchie and Johnson 2009), each management action should be planned after a full review of the main biotic interactions occurring in the ecosystem considered, so as to optimise native species conservation (Zavaleta *et al.* 2001; Bonnaud *et al.* 2009; Russell *et al.* 2009).

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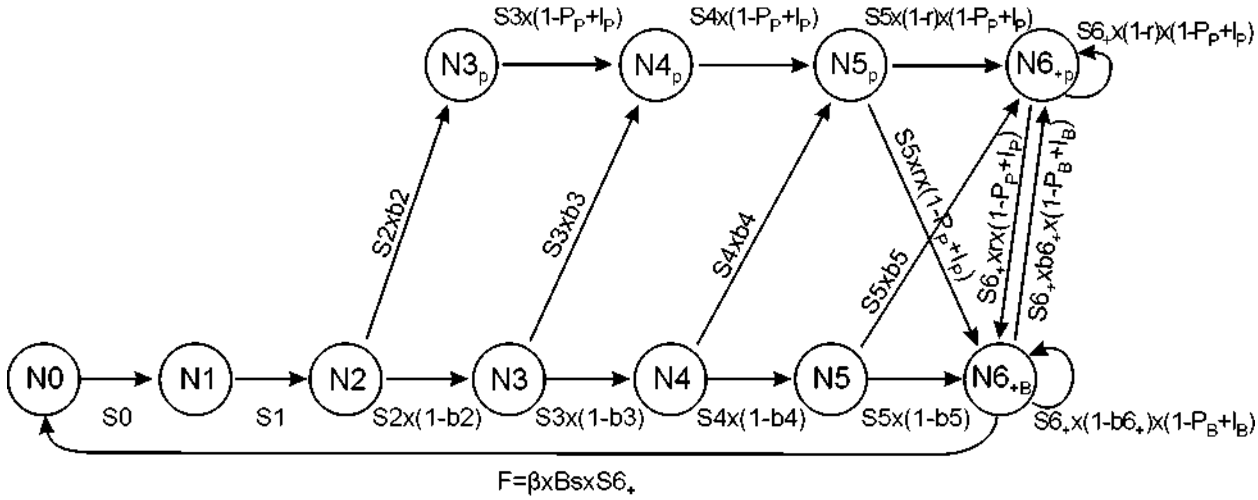
Appendix: Life-cycle representation of the population model for the yelkouan shearwater.

N0 : juvenile age-class (from fledging to age 1); *Nx* : non prospecting sub-adult of age x, Nx_P : prospecting sub-adult of age x, $N6_{+B}$: breeding adult age-class, $N6_{+P}$: prospecting adult age-class, Sx: survival of stage x, bx: percentage of birds of stage x prospecting the colony without breeding, Bs: breeding success, β: sex ratio, F: fecundity, P_B: predation rate on breeding birds, P_P: predation rate on prospecting birds.



Demographic parameters of the Yelkouan shearwater population (based on Bonnaud *et al.* 2009). Standard deviations (s.d.) are given for mean values.

Yelkouan shearwater age-classes	Population proportions with a stable distribution	Shearwater population sizes	
		Port-Cros	Le Levant
N0	0.161	143	1035
N1	0.093	83	598
N2	0.0715	64	460
N3	0.0466	41	300
$N3_P$	0.017	15	109
N4	0.0104	9	67
$N4_P$	0.0479	43	308
N5	0.0014	1	9
$N5_P$	0.0237	21	152
$N6_{+B}$	0.4044	360	2600
$N6_{+P}$	0.1232	110	792

Parameters	Values
S0: survival of stage Juvenile[a]	0.586
S1: survival of stage 1[a]	0.781
S2: survival of stage 2[a]	0.902
S3: survival of stage 3[a]	0.930
S4: survival of stage 4[a]	0.930
S5: survival of stage 5[a]	0.930
S6+: survival of stage 6+[a]	0.930
β: sex ratio[b]	0.5
Bs: breeding success[b]	0.808 ± 0.105
b2: prospecting birds of stage 2[c]	0.267
b3: prospecting birds of stage 3[c]	0.756
b4: prospecting birds of stage 4[c]	0.911
b5: prospecting birds of stage 5[c]	0.978
b6+: prospecting birds of stage 6+[c]	0.261
r: prospecting ads - breed next year[d]	0.96 ± 0.02

Data from: [a] Perrins *et al.* 1973; Brooke, 1990; Hamilton and Moller 1995; Hunter *et al.* 2000; Ainley *et al.* 2001 (*P. puffinus*); Cuthbert *et al.* 2001; Jones 2002 (*Puffinus* sp.), [b] our study, [c]. Bradley *et al.* 1999 (*P. tenuirostris*), [d]Warham 1990

The use of volunteer hunting as a control method for feral pig populations on O'ahu, Hawai'i

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Abstract The O'ahu Army Natural Resources Program and the State of Hawai'i Department of Land and Natural Resources Division of Forestry and Wildlife Natural Areas Reserve System initiated feral pig eradication within fenced management units using volunteer hunters with hunting dogs and a staff escort. This method successfully removed a large percentage of the animals trapped within these units. This control method bridges the divide between the hunting community and programmes aimed at the conservation of natural ecosystems. The aim is to build a rapport and educate hunters about the biological resources and restoration work that the agencies are trying to accomplish. This outreach reduces potential conflicts and vandalism while encouraging new conservation partnerships. Since 1997, about 688 ha of endangered species habitat in 17 management units ranging in size from 10 to 175 ha have been fenced on O'ahu. Pigs have been eliminated from three units (93 ha) and reduced to low densities in two units (242 ha) using volunteer hunters. Planning for each hunt utilises information gathered from various aspects of the programme to ensure strategic and systematic coverage. Results of scouting, fence inspections, game cameras, and GPS dog collar tracks from previous hunts are considered in planning hunt strategy.

Keywords: eradication, fencing, Natural Area Reserve, *Sus scrofa*

INTRODUCTION

Feral pigs (*Sus scrofa*) have become a problem following their introduction to such diverse locations as Australia, the Galapagos Islands, New Zealand, Seychelles and the United States, where their direct and indirect ecological impacts are well documented (Spatz and Mueller-Dumbois 1975; Springer 1977; Singer *et al* 1984; Kroll 1985; Loope *et al*. 1988; Aplet *et al.* 1991; Vtorov 1993; Atkinson *et al* 1998; Atkinson and Atkinson 2000; Sweitzer and Van Vuren 2002). The control and eradication of feral pigs has been attempted worldwide for many years (Barret and Stone 1983; Veitch and Bell 1990; Katahira *et al*. 1993; Caley and Ottley 1995; Lombardo and Faulkner 2000; Cruz *et al.* 2005; McCann and Garcelon 2008) often using dogs to trail, bail and/or catch the animals (Barret and Stone 1983; Katahira *et al*. 1993; Caley and Ottley 1995; Lombardo and Faulkner 2000; Cruz *et al.* 2005; McCann and Garcelon 2008).

Here, we report on the effectiveness of volunteer hunters with dogs as the initial tool to eradicate feral pigs from fenced habitats on the island of O'ahu, Hawai`i, USA. Secondary tools, not recorded in detail in this paper, are snaring and possibly trapping. These "mainland island" fenced habitats, or Management Units (MUs), are designed to protect endangered species of native plants (Table 1) as stable populations through management of the taxa and their habitat. Typically, these MUs are fenced areas from which ungulates and other threats are removed or controlled.

We have compiled the results from five MUs where the O`ahu Army Natural Resources Program (OANRP) and the State of Hawai`i Department of Land and Natural Resources Natural Areas Reserve System (NARS) utilised volunteer hunters with dogs under the direction of a staff escort as the primary management tool to initiate the removal of pigs.

METHODS

Study Area

This study was conducted on O'ahu Island, Hawai`i, USA, in five separate MUs ranging in size from 10 to 175 ha and referred to as: 'Ēkahanui, Kapuna/Keawapilau, Mākaha Subunit I, 'Ōpae'ula, and Palikea (Fig. 1, Table 1).

Fencing

Two types of fencing materials were used depending on terrain (Fig 2): 1) 82 or 107 cm tall hogwire mesh, cost US$269/100 m; and 2) 1.1m high welded livestock panel, cost US$1247/100 m. The more expensive panels were

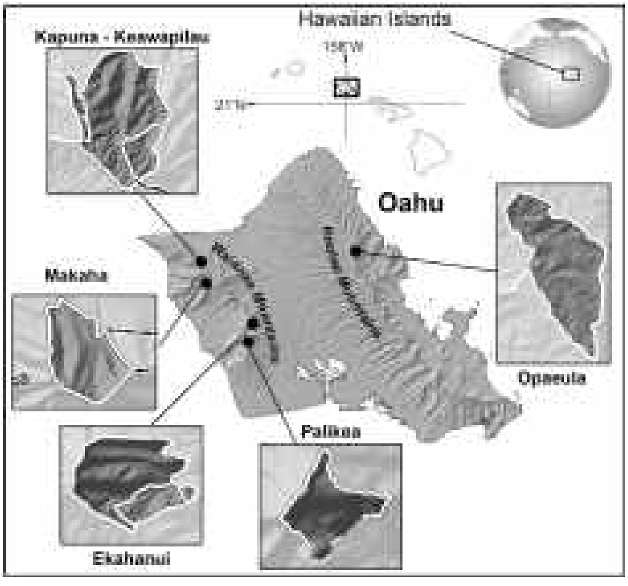

Fig. 1 O'ahu in the Hawaiian Islands with Management Units (MUs) highlighted.

Pages 402-406 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

Table 1 Description of each Management Unit (MU) including the number of listed threatened and endangered species.

Management Unit	Elevation (m)	Area (ha)	Dominant native plant species	Listed species
‘Ōpae‘ula, Ko’olau Mountains	732-823	50	*Metrosideros polymorpha*, *M. rugosa*, *Cheirodendron trigynum*, and *C. platyphyllum*	11
Mākaha Subunit I, Wai’anae Mountains	366-927	39	*Acacia koa, M. polymorpha, Diospyros* sp., and *Dicranopteris linearis*	16
Palikea, Wai’anae Mountains	878-943	10	*A. koa, M. polymorpha,* and *D. linearis*.	7
‘Ēkahanui, Wai’anae Mountains	525-954	83	*A. koa* and *M. polymorpha*	24
Kapuna/Keawapilau, Wai’anae Mountains	464-778	175	*A. koa*, *M. polymorpha*, and *D. linearis*	13

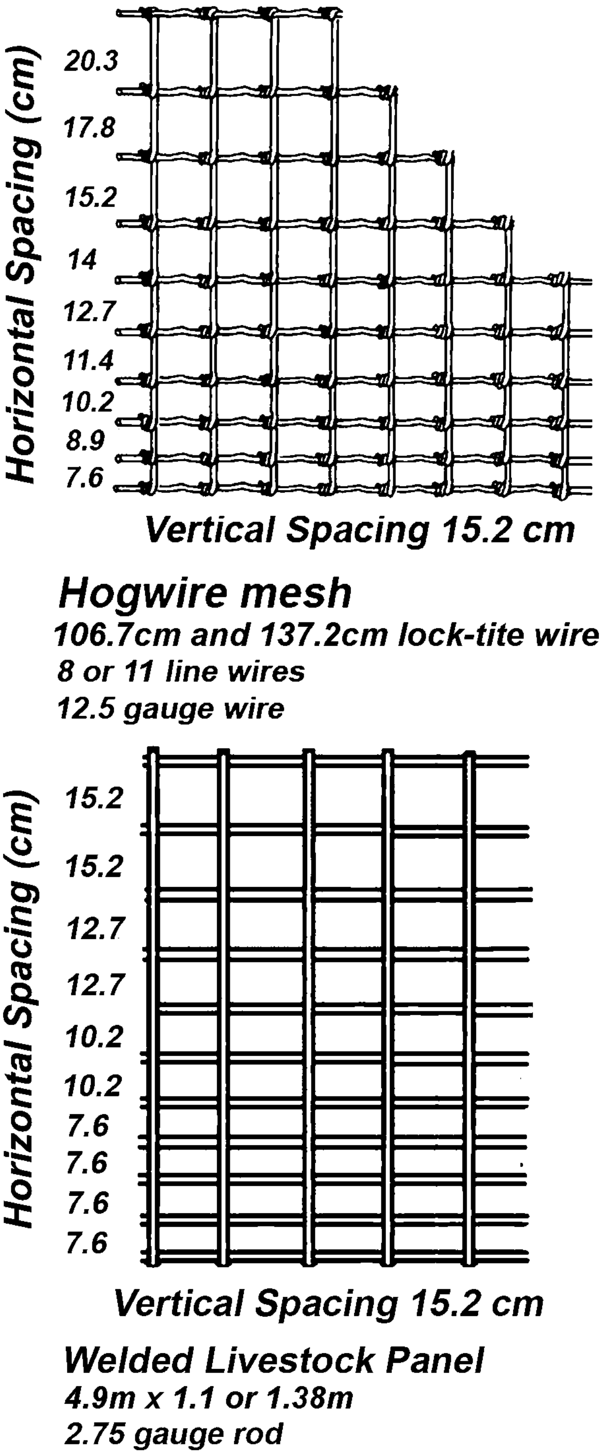

Fig 2 Details of hog-wire mesh and livestock panels.

rigid, easily transported along the cleared lines, and were cut and manipulated to fit the landscape. The taller fencing was used where there was a threat of entry by feral goats (*Capra hircus*). The fence was typically either anchored into the ground or buried along its length to prevent pigs from digging underneath.

In areas with high pig traffic outside the MU, or with very loose soils, a length of hogwire mesh was connected along the outside of the fence as a “skirt” partly up the fence and partly on the ground (Fig. 3). This was connected with hog-rings along the base of the fence from the second or third horizontal wire down to the bottom and positioned so that the smallest holes of the skirt mesh overlapped the smallest holes of the fence mesh. The “skirting” was then anchored tight to the ground. This helped slow erosion caused by foot traffic (human and ungulate) along steep sections of fence, and stopped pigs creating entrances.

Hunting with dogs

This technique involves hunters walking through an area while allowing their dogs to search for pig scent. Any pigs located are chased by the dogs and caught or bailed until hunters arrive and dispatch the animal with a knife or firearm. This technique is particularly effective for pigs that are shy of other removal techniques and in areas that contain small remnant populations. Elsewhere, contract or staff hunters were used in conjunction with other removal techniques since public hunting alone is often unsuccessful with eradication. However, in accessible areas, public hunting could be effective for the initial reduction (Barret and Stone 1983; Anderson and Stone 1993).

Volunteer hunters and staff escorts ranged from groups of two to six people and the number of dogs ranged from

Fig. 3 The skirt of hogwire mesh added to a fence of the same material.

four to 21. Most of the hunters came from communities close to the MU and had personal ties to the area. They were both sport and subsistence hunters with varying skill levels and experience.

Various dog breeds were used. Hound mixes were utilised for their ability to follow scent trails. The "catch dogs" were usually of some bull terrier mix. The quality of the dogs used varied from very experienced and well trained, to young first-year dogs at early stages of training.

Monitoring

A Garmin (Olathe, KS) Astro 220 Global Positioning System (GPS) unit and two DC 30 GPS dog tracking collars were utilised to monitor dog movements on all hunts at Kapuna/Keawapilau. The GPS collars communicate with a handheld GPS unit via Very High Frequency (VHF) radio signals. A topographic map on a 66 mm colour display (updated every 5 sec.) on the GPS unit showed the location dogs, the distance and path a dog had taken, and whether or not the dog was moving. The spatial data collected by the unit was then downloaded and converted into GIS shapefiles using ARCMAP (ESRI Redlands. CA) and the Minnesota Department of Natural Resource free DNR Garmin Application. These shapefiles were used for future hunting plans to ensure coverage of previously unchecked locations.

Two Moultrie (Alabama) Game Spy 4.0 wildlife game cameras were used sporadically and throughout the Kapuna/Keawapilau MU. Triggered by motion or infrared sensors, they provided digital photos or videos of passing wildlife 24 hours a day.

RESULTS

Details of each MU, fences, hunting effort, and results are given in Table 2.

ʻŌpaeʻula

To make this MU pig proof, skirting was applied around the whole unit. There are two stream crossings, one is at a waterfall and the second utilises a Hypalon (chlorosulfonated polyethylene synthetic rubber) sheet to block access for pigs and still allow water to pass unimpeded.

When the fence was completed, the amount of pig sign indicated that there was only one small pig left within the unit. Snares were installed by qualified staff members, but were removed after one year without catching any animals. Then hunters were flown in by helicopter. They split up into two groups and spent two days covering the unit. The sign indicated that the one remaining pig had pushed out under the Hypalon during the hunt.

Mākaha Subunit I

Skirting was applied in the steepest and most pig prone areas. Once the fence was completed, hunts with and without staff escorts were conducted over a 20-month period. Animals caught ranged in weight from 4.54 to 61.24 kg with a mean of 23.07 kg. Eight males, five females (one of which had eight embryos), and 14 infants (sex not determined) were removed. This eradicated pigs from the MU.

Palikea

Skirting was applied in the steepest and most pig prone areas. An 18.14 kg male pig was caught during the first hunt but none in the second, although a single pig was known to be present. This last pig was later caught in a snare.

ʻĒkahanui

This MU is comprised of two subunits, the first being completed in 2001 and the second in 2009. Skirting was applied in the steepest and most pig prone areas. Hunts with staff escorts began in November 2008, prior to the final completion of the fence. Three males, four females, five infants (sex not determined) and four unknowns (data not collected) were removed but no weights were recorded. The hunting was limited to three months. Snares were deployed and two more sows were captured. One animal is left within the MU.

Kapuna/Keawapilau

The Kapuna/Keawapilau MU is comprised of three subunits (I, III, IV). Subunits I and III are small and are considered free of pigs. Subunit IV is larger and was the focus of pig eradication through public hunting. Skirting was applied in the steepest and most pig prone areas. NARS staff escorted the public hunts, which commenced soon after the completion of fences in August 2008. All except three animals were taken by dogs and hunters; two were shot by staff and one was trapped. Animal weights ranged from 2.27 to 68.04 kg with a mean of 27.22 kg. Nineteen were females, six were males, and one was undocumented. Ungulate control is continuing with public assistance.

During the hunts at Kapuna/Keawapilau, GPS collars provided real-time spatial data of dog locations.

Images recovered from game cameras aided in assessing animal presence *vs.* absence and helped with identifying targeted individuals (i.e. one large boar, which was brushing off dogs and evading capture).

DISCUSSION

Across all five MUs, 60 volunteer hunters participated in 117 hunts. Beyond hunting opportunities for the public, this

Table 2 Details of each Management Unit (MU) and hunting effort.

MU	Size (ha)	Year	Length (m)	Fence Type	No of Hunts	No of Hunters	Hunter Hours	No of Pigs	Hunter hrs/ha	Person hrs/pig
ʻŌpaeʻula	50	2002	3490	Hog-wire	1	6	66	0	1.92	
Mākaha Subunit I	39	2007	2890	Hog-wire & Panel	66	26	1299.5	27	35.3	47.4
Palikea	10	2007	1506	Hog-wire & Panel	2	2	66.6	1	6.6	66.6
ʻĒkahanui	83	2009	5000	Hog-wire & Panel	26	28	777.40	16	11.96	48.6
Kapuna/Keawapilau	175	2008	6280	Hog-wire & Panel	22	24	1369	26	9.9	52.7

pig eradication method also created a working relationship between federal and state conservation organisations and a core group of pig hunters. Hunters within the community "spread the word" about management work we were accomplishing and also acted as eyes and ears in the field. In MUs that were accessible, hunters reported damage to the fences such as tree falls and blow-outs between our fence inspections. They also reported possible issues with other hunter groups that needed our resolution. We have thus been able to gain knowledge from their expertise and get a better understanding of the hunting community.

Volunteer hunter programmes required many different stakeholders. As an example, in Mākaha, we partnered with the City and County of Honolulu Board of Water Supply (landowner), Mauna ʻOlu Estates (site access), Kaʻala Farms (non-profit community organisation), and State of Hawai'i Department of Land and Natural Resources. Community meetings were held to inform the public of the fence project and gain support by local hunters. Four "good faith" hunts were conducted prior to the completion of the fence. A total of seven community hunters and four staff escorts participated for a total of 76 person hours, removing two pigs. Although a modest effort, this showed all those involved that such an operation could be successful and could reach a common goal. These early efforts led to a pig eradication programme in Mākaha that was led by volunteer hunters. Once the eradication effort was complete and trust was gained we were able to extend the hunts outside the MU in order to keep pressure off the fence.

Our data do not allow predictions of the duration and cost of volunteer hunters and dogs, an optimum number of dogs or hunters, or the efficiency of the technique. To offset costs, we limited the time and season of some hunting campaigns before applying other management techniques. This may give the hunters a sense of purpose to complete the eradication before the set date and focus effort on the animals such as piglets and pregnant/lactating females that may be more susceptible to hunting pressure. Any time limitation would be highly dependent on the principal management agency and its mission and mandates.

Keeping hunters focused on eradication was sometimes challenging. In ʻĒkahanui, Kapuna/Keawapilau and Mākaha we started with groups of hunters that were interested in conducting the campaign but as time went on, and catch rates diminished, that interest dwindled. We were left with a few groups of hunters that really understood what we were trying to do and why, wanted to help, and were determined to see successful conclusions of eradication campaigns.

Because each of the current and future MUs have unique characteristics, dog and hunter numbers may need to be modified to match each site. The breeds or numbers of dogs did not seem to be as important as their general disposition, physical conditioning and amount of hunting experience. The match of hunters and dogs to the MU also had an impact on the success of hunts. Hunters with previous experience in the area have valuable knowledge about pig movement patterns and effective hunt strategies. Limiting our search for hunters to those that have experience in these MUs should increase efficiency.

Hunting with dogs will not work for every MU. Cliffs and steep environments pose safety hazards for dogs and hunters, and the cost of ferrying hunters, dogs and staff by helicopter can become too high. Reliable access and safe workable environments are keys to the use of dogs and hunters.

Tools, such as GPS dog collars and game cameras, made hunts safer, more efficient, and more effective. Knowing the location of dogs with the GPS collars reduced the time to reach pigs that were bailed or caught. Pigs were dispatched faster, thus resulting in more successful catches, fewer injuries to dogs, and less suffering by the pigs. The collars will be standard practice from now on.

The GPS dog collar data also showed us pig movement patterns. Using the GPS receiver, we could observe routes that dogs took while chasing pigs, thus indicating where the pigs were running. With this information, interception zones such as major trail crossings, clearings, and ridgeline saddles were revealed. Hunts then involved pre-positioned gunners (staff or hunters with firearms) or individual hunters with 1-2 dogs at interception zones prior to releasing the main dog pack. For example, a pack of dogs taken into a gulch with a consistent water source encountered a pig that eluded capture. The pre-positioned gunner was able to shoot the pig which had outdistanced the dogs by approximately 100 m. It is very likely that this pig would otherwise have eluded capture.

Information gathered from the cameras, combined with observations made during hunts and other field activities, was used to inform public hunters of animal movement patterns, active zones *vs.* inactive zones, and to choose hunt strategies.

An in-house hunting dog programme or contract hunters could more effectively capture pigs per unit hunt effort. However, the frequency and timing of completion of MU fences and seasonal nature of the hunting do not warrant the costs associated with running an in-house hunting dog programme. The use of outside contractors for hunting also risks alienation of the local hunting community.

Feral pig populations continue to threaten conservation areas in Hawai'i, so having multiple control techniques will increase the efficiency of our ungulate control programmes. The use of volunteer hunters with dogs has produced positive results but is at a relatively early stage of development. We will continue to nurture relations with hunters through mutual trust, respect, and understanding. Good communication and strong relationships between conservation programmes and the hunting community are fundamental to the preservation of biological resources and the restoration work that the agencies are trying to accomplish. There has been some damage to fences by people, but we believe that potential conflicts and persistent vandalism can be averted with education and collaboration.

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Eradicating Pacific rats (*Rattus exulans*) from Nu'utele and Nu'ulua Islands, Samoa – some of the challenges of operating in the tropical Pacific

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Abstract The restoration of the small offshore islands of Nu'utele (108ha) and Nu'ulua (25ha) has long been identified as a priority for biodiversity conservation in Samoa. The first step towards restoration was the aerial spreading of brodifacoum to eradicate Pacific rats (*Rattus exulans*) in August 2009. Procedures for the eradication followed those used in New Zealand and involved technical experts from that country. Particular challenges included a tight operational time-frame (two months), technical problems magnified by the remote location, variable reliability of weather forecasting, working with the local community, and mitigating rodenticide exposure risks for the friendly ground-dove (*Gallicolumba stairi*) (IUCN: vulnerable). Solutions to these challenges are discussed as guidance for similar projects in remote island locations. Follow-up monitoring between August 2009 and March 2010 indicated that the eradication had been successful, but Pacific rats were detected on Nu'utele in May 2011. Nu'ulua has yet to be rechecked in 2011. DNA analyses are being organised to determine if these rats are survivors or re-invaders.

Keywords: Friendly ground-dove, helicopter bait spread, rodenticide, brodifacoum, communities

INTRODUCTION

The uninhabited islands of Nu'utele (108 ha) and Nu'ulua (25 ha) are in the Aleipata group 1.3 km off the eastern end of Upolu Island, Samoa (Fig. 1). The islands have long been identified as key sites for conservation (Park *et al.* 1992). They hold populations of the friendly ground-dove, tooth-billed pigeon (*Didunculus strigirostris*), coconut crabs (*Birgus latro)*, nesting hawksbill turtles (*Eretmochelys imbricata*), and nesting seabirds including red-footed booby (*Sula sula*), brown booby (*S. leucogaster*), brown noddy (*Anous stolidus*), white tern (*Gygis alba*), and great frigatebird (*Fregata minor*). Along with Namua (20 ha) and Fanuatapu (15 ha), the four Aleipata islands have eight plant species and two vegetation communities that are rare on the main islands of Upolu and Savai'i (Whistler 1984). Furthermore, Nu'utele and Nu'ulua are the only uninhabited islands large enough and far enough offshore to be considered as refuges for native species that are threatened by introduced rodents. The islands could thus play a key role in sustaining Samoan biodiversity.

There were no published records of mammals of the islands until Pacific rats (*Rattus exulans*) were trapped on Nu'utele in 1991 (Park *et al.* 1992) and Nu'ulua in 2004 (Parrish *et al.* 2004). No ship rats (*R. rattus*) or Norway rats (*R. norvegicus*) have been observed or trapped on the islands though both are on Upolu. Three field-based studies in temperate areas concluded that ship rats and Norway rats can colonise islands up to 1km offshore (Russell *et al.* 2008). The absence of Norway and ship rats from Nu'utele and Nu'ulua suggests that they are unlikely to reach the islands by swimming. Pacific rats are not recorded to swim distances greater than 100m (Atkinson 1986).

Pacific rats have probably eliminated burrow-nesting seabirds on the Aleipata islands and probably have negative effects on many of the native species still present. When Pacific rats were removed from islands in New Zealand, there were benefits for vegetation (Campbell and Atkinson 1999) and populations of birds (Pierce 2002), reptiles (Towns 1991), and invertebrates (Green *et al.* 2011). One of the aims of our project was to determine the benefits of rat removals in Samoa to encourage further eradications of rats from islands in the region.

The islands are in the communal ownership of the local people and form part of the Aleipata Marine Protected Area (MPA), established in 2004, and to which 11 villages are signatories. Restoration of the islands has been agreed to by the Aleipata District Community as one of the objectives in the MPA management plan and they have been involved in the development of the project since the outset.

In this paper we describe the methods used to plan and implement the eradication of Pacific rats from the Aleipata islands; outline the challenges faced by such projects in

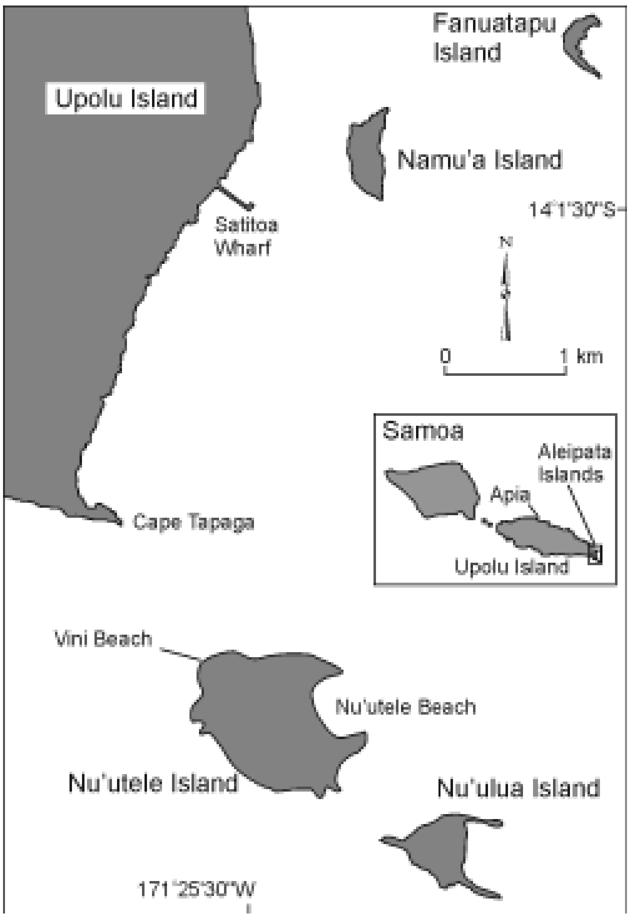

Fig. 1 Location of Nu'utele and Nu'ulua islands showing sites referred to in text.

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relatively remote tropical locations; describe the outcomes achieved; and discuss biosecurity procedures and the implications of the re-detection of rats on Nu'utele.

METHODS

Project history

The feasibility of eradicating Pacific rats from Nu'utele and Nu'ulua was first investigated in 2000. Detailed planning for the project started in 2006 with a grant to the Secretariat of the Pacific Regional Environment Programme (SPREP) from the Critical Ecosystem Partnership Fund (CEPF) through the Regional Natural Heritage Programme. At one point, the eradication was scheduled for the same year, but was postponed when there proved to be insufficient time to complete operational arrangements.

An Environmental Impact Assessment (EIA), developed for the Government of Samoa and the Aleipata District community in 2006, identified one threatened taxon at risk of poisoning: the Samoan subspecies of the friendly ground-dove (*Gallicolumba s. stairi*). Nu'utele and Nu'ulua islands are the last strongholds of this subspecies in Samoa and are crucial to its survival. The birds feed on seeds and fruit on the ground, which exposes them to the risk of poisoning if they consume fragments of rat bait. We undertook to protect the population by holding birds from Nu'utele in temporary captivity. No ground-doves were taken from Nu'ulua because of difficult access and also because ground-doves could be reintroduced there from the re-established population on Nu'utele if required. Temporary aviaries were set up at the Samoan National Parks & Reserves headquarters at Vailima near Apia.

The project was implemented by SPREP, with Tye as project manager and Butler as project advisor, in partnership with the Division of Environment & Conservation, Ministry of Natural Resources & Environment (MNRE), Samoa (Assistant CEO: Tipama'a). Planning for the project was scrutinised by the New Zealand Department of Conservation (DOC) Islands Eradication Advisory Group (IEAG), who reviewed its operational plan, and an experienced DOC staff member (Wylie) was provided as the Aerial Drop Adviser for the aerial operation. The project was supported technically by the Pacific Invasives Initiative (PII), which adopted it as a 'demonstration project' for the region. Wellington and Auckland Zoos and an individual volunteer assisted with the ground-dove captive holding operation.

Rat eradication

Eradication methodology was based on successful operations in New Zealand and other island groups using Pestoff Rodent Bait 20R 10 mm extruded baits containing 20 ppm brodifacoum (Animal Control Products, Wanganui, New Zealand). Baits were dyed green, contained no bitrex, and were supplied in 25 kg bags with 40 bags to a pallet making an overall weight of 1050 kg per pallet. Pallets were shrink-wrapped and shipped from New Zealand in a container with black plastic fixed internally to the roof to reduce condensation. The container was transported to near the helicopter loading site and bags of bait transported to the bucket on the back of a utility vehicle.

Baits were spread using a spreader bucket slung below a Robinson R44 helicopter flown by North Shore Helicopters (NSH). Six tonnes were ordered to cover two planned drops at 10-day intervals on each island plus 1.5 tonnes for contingency bait application. Bait drops were planned for periods when weather forecasts predicted at least three nights without rain, to ensure optimal exposure of the baits to rats. Flight lines were checked using digital GPS (TracMap).

Rats were trapped on both islands shortly before the operation and 19 tails samples collected from Nu'ulua and 7 from Nu'utele. These would be used for DNA analysis in the event that rats were found on the islands after the operation.

Ground-dove capture

Two expeditions to Nu'utele in 2006 developed capture and handling techniques for ground-doves. During the first field trip, attempts were made with a line of people to 'herd' birds into mist nets. This technique has worked elsewhere in the Pacific, but proved unproductive on Nu'utele, where the only ground-dove seen was not caught. The second trip used teams of two-three people, who quietly set up nets and left them for several hours. This group had over 20 encounters with birds. Although only one ground-dove was caught, the high encounter rate confirmed the potential efficacy of this technique.

Ground-doves needed to be caught as close to the time of rat baiting as possible to minimise the time birds were in captivity, yet the capture process could not hold up the spread of baits. Captures were scheduled for two weeks with a further week before the scheduled first spread of baits. However, delays meant that the intervening week was lost and captures of ground-doves began two weeks before the first bait drop.

A target of 10-12 birds was identified from assessments of their population on Nu'utele and the likely ease of capture. Once this number was captured, the rat baiting would proceed. A failure to capture this number would require re-assessment.

Two periods of netting were undertaken on Nu'utele on 21-23 and 27-30 July at Vini and Nu'utele beach flats (Fig. 1). The nets were visited every 1 – 1½ hours and birds were removed. The nets at Vini were left set overnight. Nets that required greater travel distances were shut down each evening. Three teams of 2-3 people worked with several nets, most of which were 60 mm mesh size (two were 40 mm). Ground-doves were transported to the campsite at Vini in cloth carry-bags and placed into small holding cages made of nursery shade-cloth. The birds were weighed and those captured on the second visit were banded. The birds were held in the cages for up to two nights before transfer by boat and car to Vailima.

Holding aviaries were built from shade cloth within a rat-proof area contained by welded wire mesh. Birds were housed as groups of two or three of the same sex and fed a Topflite seed mix for doves imported from New Zealand and containing white and Japanese millet, buckwheat, and sorghum. Twice each week, the birds were given the vitamin and mineral supplement Ornithon in their water (see also McCulloch and Collen 2009).

The survival of ground-doves left on the island would be assessed by the frequency of sightings of banded and un-banded birds after the operation.

Monitoring outcomes

Monitoring established to measure the outcomes of removing the rats included bird counts in July 2009 on a transect on Nu'utele and repeated in 2010, and lizard surveys on both islands in June 2009, December 2009 and August 2010. Photo points were also established.

RESULTS

Rat eradication

The first baits were spread on 15 August 2009. The second application of baits began on 22 August but was abandoned due to equipment failure after 80 ha of Nu'utele had been covered. This partial application was also compromised by subsequent rain. The final complete application of baits was on 26 August. The average application rate for the first drop was 12.5 kg/ha with coastal areas on both islands receiving a further 10 kg/ha. During the second, partial application 80 ha of Nu'utele (south-eastern c.76% of the island) received 8.3 kg/ha. The final average application rate on Nu'utele was 7.6kg/ha with an extra c.10.6kg/ha on the coastal areas and on Nu'ulua the rate was 11.6kg/ha. A cave at the base of cliffs on Nu'utele was hand-baited because it could not be reach from the air.

Ground-dove rescue

Twenty-six ground-doves, comprising nine adult males, one juvenile male, 15 adult females and one juvenile female, were taken into captivity. Four birds came from Nu'utele Beach, one from the hill track and the remaining 21 were from Vini Beach (Fig. 1). One net that was set on flat ground near the middle of Vini Beach, and adjacent to the base of the hill, caught 15 ground-doves.

Three ground-doves died in captivity, all in the first few days after transfer, and one was euthanased after a banding mishap. Twenty-two ground-doves were released on Nu'utele on 17 September, 22 days after the last drop. The released ground-doves were in good health after 49-56 days in captivity.

CHALLENGES

Prior to the operation

Issues with funding and project management structure

CEPF funding for the project was approved on 27 April 2009, but the funds were not received by SPREP until 2 June, only two months before the first scheduled poison drop. Since payments could not be made for baits and shipping until the funds were released, the project came within a few days of failing to meet the bait manufacturer's delivery deadlines.

Funding criteria and the history of local staff association with the project produced a problematic project management structure. SPREP managed the project as the Samoan Government were not eligible to receive the funding, but their project manager and adviser were unable to work on the project full time. In addition, between 2004 and 2009, five people were responsible for managing the Samoan Government's input to the project, including three changes to the Project Manager within MNRE during the ten weeks before the first drop. One departing manager had received significant training for rat and ground-dove aspects of the project. Another officer involved with the project since 2005 was transferred to a different section within the Ministry three weeks before the drop and was not able to be involved. The lack of full-time project management, together with these changes in Government personnel, made it very hard to maintain project momentum and ensure that tasks were completed on schedule. SPREP and DOC participants then had to take on management roles outside their advisory functions.

Helicopter contract and loading site

For several months before the operation, negotiations were held with an apparently suitable helicopter company after signing the contract. They had undertaken similar drops in Fiji, received training from a New Zealand pilot highly experienced in island eradications, held a licence to operate in Samoa and offered a price well below budget as costs could be shared with other work scheduled in the country. In late June the company advised that their helicopter needed overhaul and would not be available until October, too late for the operational window of June-September. A second company, NSH in New Zealand, was in the process of gaining a licence to operate in Samoa and provided a competitive quote. After delays while a third quote was obtained to meet the donor's requirements, approval to use NSH was finally received on 18 July. A contract was signed on 23 July, leaving little lead-in time before the first spread of baits due in the week beginning 3 August. The company was selected partly because they had an experienced pilot and back-up person who had also worked with spreader buckets.

As planning progressed, the loading site at Satitoa wharf on Upolu had to be changed after redevelopment made the previously identified one unsuitable. The Samoa Port Authority made an alternative loading site available.

During the operation

Weather forecasting

The operation was planned for early August - the middle of the 'dry' season - to minimise the chance of rain. The dry season of 2009 was wetter than normal in response to an El Niño event. Combined with forecasts of limited reliability for the local area, identifying a period with little chance of rainfall (<10mm total) over three nights became a challenge.

We used forecasting information supplied by the Samoa Meteorology Division, MNRE, the Fiji Meteorological Service, the Weather Service Office at Pago-Pago, American Samoa, and international web-sites offering long-range forecasts for Apia, particularly www.weather-forecast.com/locations/Apia/forecasts/latest because it estimated rainfall amounts.

The Meteorology Division provided specific 7-day forecasts twice a day for rainfall and wind speed at Nu'utele through a NOAA system, which estimates weather conditions for any location based on a Latitude/Longitude reference. However, these rainfall forecasts proved to be inaccurate. The Meteorology Division also provided links to weather satellite images and maps from the Australian Meteorological Service showing predicted rainfall patterns over a short time period. An automatic weather station at Cape Tapaga (Fig.1) provided records of rainfall after each bait drop.

The first bait drop was scheduled for 12 August but was postponed when up to 100 mm of rain was forecast over the following two days. Although rain continued to be forecast over the next two days none fell. Based on predictions for improving weather (3mm of rain on the 15th), followed by deterioration after 18 August (>30mm over 3 days), the first drop was re-scheduled for 15 August. Following the first drop, 6.25 mm of rain was recorded between midnight and 10 am and the next two nights were dry. These conditions met the <10mm criterion.

Forecasts before the second drop predicted 0.1 mm of rain on the third day following. However 100mm fell over the next three days and nights so this drop would have been ineffective if it had been completed.

The final drop followed a suitable weather forecast after which there were five nights with negligible rain. However, both completed drops were undertaken in windy conditions, the first having occasional gusts of up to 25 knots. Nonetheless, flight lines downloaded from the GPS indicated that these conditions should not have affected bait coverage.

Helicopter operation including equipment testing and calibration

Because of tight timeframes after signing the contractor, there was insufficient time to calibrate the spreader bucket with non-toxic bait. NSH advised that the bucket had been tested many times for previous operations and that all aperture settings were already recorded. However, although the pilot and bucket were sourced from one company, the helicopter and GPS system originated from another.

The equipment expected in Apia on 6 August was further delayed when the freighter was diverted to another port en route and did not arrive in Apia until Saturday 8 August. The port does not operate on Sundays, and Monday was a public holiday so nothing was unloaded until 11 August. The helicopter and GPS system were tested on the 13 August and the spreader bucket was also tested, although unattached to the aircraft.

The delays meant that the agreed pilot and back-up both returned to New Zealand and were replaced by a new pilot with agricultural flying experience, but less familiarity with the GPS and spreader bucket systems.

Technical issues encountered during the first drop were: 1) an incorrect shackle attachment between the bucket and the helicopter had to be replaced as it had been set up for a different model; 2) the light panel on the TracMap navigation system that provides the pilot with course information stopped working; 3) the bait spread adjustment aperture at the base of the bucket closed itself during flight, was fixed with tape and cable ties, but had to be checked every time the pilot landed; 4) after the fourth load, the bucket spinner engine needed re-fuelling after unexpectedly heavy fuel use, jammed on re-starting, needed to be disassembled, would then only run with the air filter removed, and this further increased fuel consumption and the need for re-filling; 5) the TracMap flashdrive was left in Apia so it was not possible to download flight lines, check for gaps, and re-fly them at the time; and 6) a day after the drop, a patch on the cliff side of Nu'utele Island was found to have been left un-baited. The area would have been re-flown during the first drop had it been possible to monitor bait application, but was covered in the subsequent drops.

The bucket continued to give problems until the engine finally seized part way through the second drop. A replacement engine was flown from New Zealand for the third drop.

SUCCESSES

Baits

Bait transfer and storage were not beset by problems. We followed ACP recommendations and every few days opened the bait container door during the day (if fine) and closed it at night. The shrink wrap was left on the pallets until the first drop as there was no sign of condensation and the bags remained moisture-free while in storage.

Liaison with local communities

One family was recognised by local communities as the main users/owners of Nu'utele where they maintained two *fale* (open huts) and occasionally worked a small plantation of bananas or taro. Nu'ulua is not used by any families due to the difficulty of access. However, the whole district had an interest in events on the islands and the project maintained close liaison with the MPA District Committee.

The community was first asked to support the concept of rat eradication in 2000 and was involved in all subsequent project discussions. The community was then asked to re-endorse the project as it progressed and as difficulties arose. Liaison with the community was largely undertaken by the marine section of MNRE and the community remained supportive throughout. During implementation, members of the MPA District Committee observed the bait drops and villagers were employed to load the bucket with bait. Continuing community discussions are planned including debriefs for the MPA District Committee, an education programme for local schools, and further biosecurity training and implementation.

Health and Safety

On the morning of the first bait drop, the pilot and one of us (Wylie) provided a safety briefing for the villagers employed to load the bucket with MNRE staff providing translation into Samoan. The briefing was complicated by the pilot's unfamiliarity with the site and lack of briefing about how bait loading would proceed. However, the loading crew functioned well throughout and wore all protective clothing supplied, despite the very hot conditions.

The operation was safely observed by staff from MNRE, other conservation agencies, members of the MPA committee and other interested individuals.

OUTCOMES

Pacific rats

Nu'utele Island has been visited at least eight times since the spread of rat baits and Nu'ulua twice.

Both islands were visited in December 2009 and August 2010 during surveys to assess changes in lizard populations. Four-person teams undertook day and night surveys and set out 500 glue traps on each visit at a variety of locations. No glue traps had rat hair compared with 75% of traps showing evidence of rats in a pre-operational lizard survey in June 2009 (R. Fisher, unpublished data). However one of December's team subsequently reported seeing a rat at Vini Beach. This was partly discounted as it was not reported at the time, though two lines of traps were set up there in February 2010 and caught nothing.

A specific survey for rats was carried out on Nu'utele in March 2010. Poor weather prevented access to Nu'ulua. Kill traps, cage traps, bait stations, wax tags and tracking tunnels were deployed for a week on grids or transects covering different parts of the island. Fallen fruit was checked for any signs of chewing. No rats or rat sign were detected.

In late 2010, teams studying the invasive yellow crazy ant (*Anoplolepis gracilipes*) on Nu'utele recorded no rats. However in May 2011 one of these teams saw a rat on Nu'utele towards the top of the climb up from Vini beach. A follow-up survey in July caught 8 Pacific rats in this area and two at the northern end of Vini Beach. A brief trapping session on the coast of Upolu opposite the islands caught one Pacific rat, three Norway rats and two ship rats.

Friendly ground-doves and other native biodiversity

Within the first few weeks after the spread of baits, at least six ground-doves were seen on Nu'ulua and banded (released) and un-banded birds were seen on Nu'utele. These observations show that some of the ground-doves on both islands survived the poison drops. All subsequent expeditions to Nu'utele have reported greater frequencies of ground-dove sightings than before the operation.

The monitoring programme, which includes 5-minute bird counts, photo points, and lizard surveys, is continuing and results are not yet available. However quite dramatic increases in the ground cover of seedlings is apparent in many areas of Nu'utele.

BIOSECURITY

Biosecurity training to prevent rats reaching the islands was organised by PII in New Zealand, and attended by the Assistant CEO, MNRE, with overall responsibility for ongoing Government participation in the MPA, a representative of the family that use Nu'utele, and by two members of the MPA Committee. This training was cut short due to a tsunami affecting Samoa and completed in February 2010.

The tsunami reduced the risk of rats being accidentally taken back to the islands by boat for it damaged the nearest wharf and destroyed most fishing boats in the district. However, it could have increased the risk of an incursion through rats 'rafting' to the islands on debris which reached Nu'utele from Upolu.

For over a year after the operation, the MPA Committee inspected the equipment and supplies of all expeditions visiting the islands but this practice lapsed by mid-2011. Traps and bait stations have been placed on Vini and Nu'utele beach flats, Nu'utele Island and on the only beach flat on Nu'ulua Island, but not visited and re-baited as often as desirable.

DISCUSSION

The spreading of a prescribed amount of rodenticide bait was eventually achieved on both islands. However the subsequent detection of rats on Nu'utele is clearly a significant setback. There is little to be gained from speculation about whether these are survivors or re-invaders and it is hoped that this question can be answered shortly from DNA analyses. If the initial eradication did fail it is clear that this was not a widespread failure and much of Nu'utele is considered to have been free of rats up to now.

It was not possible to reach Nu'ulua during either the March 2010 or July 2011 rat surveys so the current situation there is unknown. Efforts are currently being made to reach the island using helicopters which are temporarily stationed in Samoa for filming.

The number of ground-doves captured before the baiting operation exceeded our expectations. The efforts of overseas experts and local staff ensured that ground-dove mortalities were restricted to four birds, which is not exceptional for a programme of this kind. Much was also learned about keeping this species in captivity. Biometric data and DNA samples collected from each captured bird will prove valuable to future conservation efforts.

Despite considerable planning effort, many last-minute problems with the rat eradication campaign could have jeopardised its success. Most such issues were not completely unexpected, but some were exacerbated by the short period between the receipt of funding and the operation. Funders and project planners may need to allow for the long lead-in time required for such operations.

The turnover of Government staff involved in the project is not unusual in the Pacific, although the changes immediately before the operation were exceptional. Ideally, the same Government official would have a key management role throughout the eradication project and then supervised the subsequent biosecurity and restoration work. This situation may be unlikely in Small Island Developing States. In our project, some continuity was provided through the involvement of one of the owners of the island (a former MNRE staff person) and the project management team.

Weather forecasting can also be problematical in small island countries. The best strategy is to choose times of year with minimal rainfall and to purchase enough bait to do additional drops if the first ones are washed out.

An open tender process might be the best approach for helicopter support, but was unachievable for the Aleipata project. Pilot experience also requires consideration, and ideally should be made a contractual requirement. Obtaining such agreements may be difficult in remote locations, particularly if schedules may be uncertain.

Some of the technical problems associated with the helicopter might have been avoided if more time had been available for testing, especially since spreader buckets are a 'weak link' that has generated problems elsewhere (D. Merton pers. comm.). Ideally, two buckets should be on site, but this was not possible in Samoa because of cost and problems of availability. Where a second bucket cannot be provided, a good range of spare parts must be held, including if possible a spare engine to drive the spinner.

A biosecurity programme should be in place before an eradication proceeds. In this case, the two months between securing the funds and carrying out the operation were fully spent organising the poison drops. Sufficient lead-in time should be planned to allow biosecurity measures to be completed beforehand.

Funding delays for the Aleipata project were sufficient to postpone it for another year. But cancellation two months out would have been a very hard call to make because of the years taken to put key elements in place, namely: funding, Government and community support, and overseas individuals and agencies with time and resources committed. Once the decision was made to proceed, commitments were immediately entered into for the purchase of bait and travel for advisers. This meant that any last minute postponement would have led to the loss of significant funds, credibility, and support.

Further discussion is needed on how to match the thorough planning and checking that are features of successful operations in larger countries with the situation in the Pacific where many challenges can arise. There is clearly a higher risk of failure operating in a remote location but what level of risk is acceptable? What is the appropriate balance between the use of outside experts - as a key way of minimising risk, and involving local staff - as a means of building local capacity?

However well-planned and structured a project is, there seems little doubt that the ultimate key to operational success is the combined skills and commitment of those on the ground.

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Density estimates and detection models inform stoat (*Mustela erminea*) eradication on Resolution Island, New Zealand

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Abstract Resolution Island (20,800 ha) in Fiordland, New Zealand, has long held great potential as a sanctuary for the protection and reintroduction of highly threatened bird species. In 2008, the New Zealand Department of Conservation initiated a programme to eradicate wild stoats (*Mustela erminea*) from Resolution Island. Following the establishment of a trapping network, but prior to the traps being set, hair-snagging devices were deployed on approximately one quarter of the island, in order to obtain an independent estimate of population density. Stoat hair samples were collected from devices approximately daily over a 10-day period. DNA was extracted from 117 hair samples, and resulting genotypes were analysed using the spatially explicit mark-recapture software DENSITY, which provided a population density estimate for the study area of 0.48 stoats km^{-2} (95% CL 0.31 – 0.74; CV 23%). Hair tubes underestimated the 'minimum number alive' population density calculated from the number of stoats subsequently captured in kill-traps (an estimate of 1.4 stoats km^{-2}) but provided precise information on detection parameters. They also gave an independent measure of initial trapping success with 21 out of 22 stoats detected in tubes being subsequently caught in traps. The above data in a Lincoln-Peterson index, with hair samples as the mark and trap samples as the recapture, gave a population estimate slightly above the actual number trapped. In a preliminary analysis, we modelled trap-capture data in a Bayesian framework and estimated that the probability of stoats persisting would be <1% after 10 consecutive checks with no captures. These models also yield a population slightly higher than the number of animals actually caught. We conclude that DOC150 traps were efficient at detecting stoats, but trapping stoats to extinction on Resolution Island will not be achieved in the near future and that initial trap spacing may have contributed to this.

Keywords: Bayesian modelling, detection parameters, Fiordland, genotyping, *Mustela erminea*, Resolution Island, restoration, stoat

INTRODUCTION

Stoats (*Mustela erminea*) are an invasive alien predator implicated in the historical and continued decline of many highly threatened bird species in New Zealand such as kiwi (*Apteryx* spp.), kaka (*Nestor meridionalis*), mohua (*Mohoua ochrocephala*), takahe (*Porphyrio hochstetteri*), and blue duck (*Hymenolaimus malacorhynchos*)(King and Murphy 2005). One way to effectively manage the threats posed by stoats is to eradicate them from offshore islands, thereby creating 'island sanctuaries'.

In 2002, following successful invasive mammal eradications on other New Zealand islands and around the world (Simberloff 2001; Veitch and Clout 2002; Howald *et al*. 2007), the New Zealand Department of Conservation (DOC) initiated a plan to eradicate stoats from Resolution Island (detailed in Edge *et al*. 2011). The island (ca 20,800 ha) is the largest of Fiordland's near-shore islands. The only introduced mammals on the island are stoats, mice (*Mus musculus*), and deer (*Cervus elaphus*). The eradication of stoats would create the largest island sanctuary in New Zealand for highly threatened bird species such as the kakapo and those with large home range requirements such as kiwi and kokako (*Callaeas cinerea wilsoni*) (McMurtrie *et al*. 2008).

The size and remote location of Resolution Island have made this attempt extremely challenging. Furthermore, at its narrowest point the island is only 520m offshore. Stoats are trapped on the adjacent mainland coast, but the narrow channel is well within their swimming capabilities (Taylor and Tilley 1984). Although design of the current operation involved scaling up from previous campaigns on smaller islands (Edge *et al*. 2011), it was not known how the capacity of stoats to reinvade might compromise the eradication attempt (Elliott *et al*. 2010).

The planned eradication of stoats from Resolution Island provided an important opportunity to apply learning from earlier eradication campaigns and to fit these and the current research into an adaptive management framework. Key questions revolved around the number of stoats on the island prior to control, and the number of stoats remaining following the initiation of control. Independent estimates of the initial population can be obtained by: 1) using microsatellite DNA analysis of hair to identify individuals (Foran *et al*. 1997) and analysing these data in a mark-recapture framework; and 2) using a Bayesian analysis of the kill-trapping data. Microsatellite DNA analysis can also be used to determine the genetic relatedness between island and mainland populations (McMurtrie *et al*. 2011), which is important for identifying the origin of animals that are caught during later phases of the eradication programme.

In this paper, we present results from research on Resolution Island which aimed to: 1) determine the initial population size and density using mark-recapture models based on genotyped hair samples (Lincoln Peterson index, Seber 1982; Program DENSITY, Efford *et al*. 2004) and a Bayesian model for the trap-capture data; 2) estimate spatial detection parameters (capture probability and home range width) of the stoat population prior to eradication using the same molecular data; and 3) provide an estimate of the search effort necessary to declare eradication success.

METHODS

Study area

Resolution Island in Fiordland, New Zealand (45° 41.4´S, 166° 41.5´E) reaches 1069 m (Fig. 1). The vegetation is a mix of southern beech (*Nothofagus menziesii* and *N. solandri* var. *cliffortioides*) and podocarp-broadleaf forest dominated by kamahi (*Weinmannia racemosa*) and rimu (*Dacrydium cupressinum*); manuka (*Leptospermum scoparium*) shrublands; tussock grasslands dominated by *Chionocloa acicularis*; and small areas of wetland, coastal scrub and fellfield vegetation (Ledgard and Rance 2008). The climate is cool temperate, with mean annual temperature of c. 10° C, and annual rainfall of c. 4000 mm spread evenly throughout the year (Bayliss *et al*. 1963).

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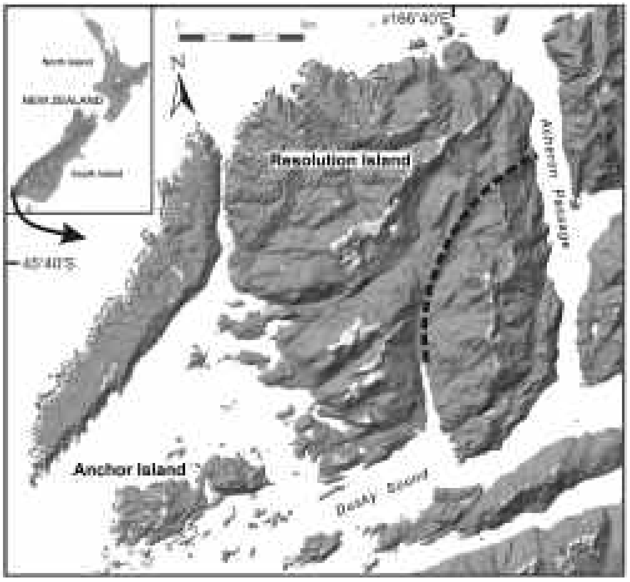

Fig. 1 Resolution Island in Fiordland, New Zealand. The study area used for non-invasive sampling of DNA from the stoat population prior to initial knock-down is south-east of the dotted line.

Pre-baiting and kill-trapping

The trapping regime for stoats on Resolution Island was similar to that for Secretary Island (McMurtrie *et al.* 2011), with a network of tracks covering the island and DOC150 traps spaced at c. 100 m intervals on each track. Each trap was placed inside a protective wooden tunnel (400 mm × 150 mm × 200 mm) and the goal was to have no point on the island more than 700 m from a trap. Track spacing was approximately equal across the whole island. Pre-baiting with eggs and meat was conducted twice, on 20 May and 24 June 2008 and kill-traps were set, checked and re-baited twice over two 3-day periods from 15–24 July and once more during 5–12 August 2008. This resulted in a total of three initial trapping sessions over 20 days during the "knockdown" phase.

Genotyping

Prior to the initial knockdown, and between pre-baiting sessions (4–13 June 2008), we obtained DNA from hair follicles (Foran *et al.* 1997) of stoats on c. 5900 ha (28% of the island; Fig. 1) using hair-snagging tubes (Duckworth *et al.* 2006). A total of 208 sections of PVC drainpipe (250mm length x 40 mm diameter) were placed 1–2 m from every second wooden trap box (i.e. every c. 200 m). Each tube contained two rubber bands stretched through slots at each end, pasted with a 50:50 mixture of trapper glue (Bell Laboratories, Wisconsin, USA) and toluene ultimAR (Mallinckrodt chemicals) and baited with a small piece of fresh rabbit meat, secured to the ground with a wire hook. Tubes were checked and re-baited when weather conditions permitted; on average three out of every four days to provide a 1-session closed population estimate with five occasions over a 10-day period. Tubes containing hair samples were replaced with a fresh tube. The hairs obtained were left attached to each rubber band, which was snipped off using forceps and scissors. Each length of rubber band with hairs attached was then wrapped in filter paper and samples sent to EcoGene™ (Auckland, NZ) for DNA extraction. Tissue samples (tail tips) were also collected from stoats captured in kill-traps and those samples that came from the study area were included in the Lincoln Peterson mark-recapture estimate.

In the laboratory, 50 mg of muscle tissue and caudal skin were removed from the tail tips and DNA was isolated using a Bio-Rad AquaPure Genomic Tissue Kit (Cat# 732-6343) following the manufacturer's protocol. DNA extraction from hair samples used a modified protocol following Walsh *et al.* (1991). Hair follicles were placed in an Eppendorf tube containing 100 µl of extraction buffer (5% chelex 10 mM Tris, 0.1 mM EDTA), followed by an addition of 1 µl Proteinase K (20 mg/ml) and 2.7 µl of 1 M DTT. Samples were incubated at 56°C for 2 h. A further 1 µl of Proteinase K was added and samples incubated an additional 2 h at 56°C, tapping occasionally. Samples were then boiled for 8 min, vortexed at high speed for 15 s and centrifuged (13,000 rpm) at room temperature for 3 min. Supernatants were transferred to new tubes with a wide-bore pipette tip, and stored at −20 °C.

Microsatellite amplification and genotyping across 16 variable microsatellite loci followed McMurtrie *et al.* (2011). Evidence for allelic drop-out, scoring error due to stutter, and presence and frequency of any null alleles were assessed with MICRO-CHECKER (Oosterhout *et al.* 2004). Genotyping was carried out using a step-wise protocol of exclusion that has been shown elsewhere to ensure rigorous and conservative determination of identity (Paetkau 2003; Weaver *et al.* 2005). We required a perfect match between the two amplifications in order to accept each genotype and to eliminate PCR errors resulting in either allelic drop-out or false alleles. Any samples that differed by one locus were checked for potential scoring or amplification errors (Paetkau 2003). If these differences were not able to be explained by errors in scoring/typing, samples were then subjected to a further round of PCR and scoring (Poole *et al.* 2001; Mowat and Paetkau 2002). Samples that were not able to be accurately genotyped for the majority of loci were rejected from the analysis.

We used the software package GIMLET (v. 1.3.3; Valiére 2002) to estimate P_{ID} and $P_{ID\text{-}sib}$ among full siblings as that provides an upper limit to the probability that pairs of individuals will share genotypes (Taberlet and Luikart 1999).

DATA ANALYSES

Stoat density on the south-eastern part of Resolution Island was estimated in two ways. First, by spatially-explicit capture-recapture in Program DENSITY (Maximum Likelihood method) (Efford *et al.* 2004) using the individual genotypes identified with DNA extracted from hair follicles. Estimating population densities (D) using DENSITY also enabled us to calculate two spatial detection parameters: 1) g0, which is per-night probability of capture at the centre of the home range, and 2) σ, which is the spatial scale over which the probability of capture declines with distance from the home range centre. The precision of the estimates of D, g0, and σ was measured using the coefficient of variation (CV); the standard deviation of an estimate divided by the estimate. Secondly, we used the total number of stoats caught from the three initial trapping sessions on the south-eastern part of the island to estimate the minimum density on the island. These data were also used to calculate initial population size (N) and the probability of capturing each stoat (θ) with the deployed traps as follows:

$$y \sim binomial(\theta, N).$$

$$\theta = 1 - \exp(-\rho * Effort)$$

where ρ is the rate parameter describing the relationship between number of sessions (Effort) and detection probability, θ. In this analysis we did not attempt to incorporate heterogeneity of detection in males and females. We then used Bayes theorem and the relationship between trapping effort and detection probability to predict the probability of stoat persistence given no detection (Ramsey *et al.* 2009).

We also used the total number of stoats caught from the three initial trapping sessions across the whole island to estimate the minimum density of stoats on Resolution Island.

Areas were calculated using ARCGIS (ESRI, Redlands California, USA).

RESULTS

Genotyping

Of 191 hair samples and 112 tissue samples obtained, 117 hair samples and all tissue samples were successfully genotyped for all 16 loci. Where DNA genotyping was not possible, most were <5 hairs and DNA yield was subsequently low. For these samples, either PCR amplification was not possible for any loci, or it was infrequent and all loci could not be reliably genotyped. The P_{ID} within the population across all loci was 0.097% and the $P_{ID\text{-}sib}$ of 4.4 x 10^{-5} was well below the 1% threshold. No identical genotypes were obtained amongst the tissue samples. There was no evidence of allele dropout or scoring error due to stutter. One locus (Mer041) exhibited some evidence of a null allele; however, because it would not affect the ability to differentiate individuals, this allele was not removed from the analysis. Given these results, identical genotypes within different individuals from this population were extremely unlikely and it is reasonable to conclude that hair samples with identical genotypes are from the same individual.

Stoat captures in kill-traps and hair tubes

Two hundred and ninety stoats were caught in kill-traps during the knockdown phase of trapping (Table 1) giving an initial minimum population estimate of 1.4 stoats km^{-2} across the island. More females than males were caught in all trapping sessions. The overall ratio of female to male stoats was >3:1 and differed significantly from 1:1 (exact binomial test; P=0.002). Most stoats (75%) were caught in the first 3-day trapping session and were caught across the whole island, in all habitat types and altitudes. In the study area, 81 stoats were captured in traps and this also equated to 1.4 stoats km^{-2} (Table 1).

Table 1 Number of stoats caught during the 'knockdown phase' of trapping on Resolution Island. Traps were pre-baited twice, set and checked twice over two, 3-day cycles (July) then checked again 14 days later in August; and detected by hair tubes within the study area.

Trapping (whole is.)	Fem	Male	Unkn	Total	Density[1]
Session 1 (July)	157	61	1	219	
Session 2 (July)	35	4	0	39	
Session 3 (August)	32	0	0	32	
Total	224	65	1	290	1.39
Trapping (study area)	64	17	0	81	1.37
Hair tubes					
Unique individuals	13	8	1	22	
Recaptures	9	7	0	16	

[1]Density estimates were derived from trap catch data divided by the sampling area (whole island, or study area only).

Twenty-two individual stoats were identified from the 117 hair samples. Twenty one of the 22 individuals (95%) identified in the hair tubes were subsequently captured in kill-traps. The ratio of female:male stoats detected in tube trap samples was 1.6:1 and not significantly different from an equal ratio (P=0.286). The median number of detected tube entries per stoat was 2.5 (range 1–27).

Population density and detection probability

The capture-recapture data (Table 1) gave an initial estimate of population density D in the south-eastern part of the island of 0.48 stoats km^{-2} (95% CL 0.31 – 0.74; CV 23%). The sampled stoat population had a g0 estimate of 0.13 day^{-1} (95% CL 0.07 – 0.22; CV 31%), and σ was 397 m (95% CL 322-489; CV 11%). In other words, a stoat had a per-night probability of being captured in a tube at the centre of its home range of approximately 13%, and a home range radius of 486 m (half of 2.45×σ; Efford *et al.* 2004). The Lincoln-Peterson index gave an estimated population in the study area of 85 stoats, which is slightly higher than the number actually caught.

Modelling the abundance of stoats using the kill-trapping data gave an estimate of N=94 stoats on the south-

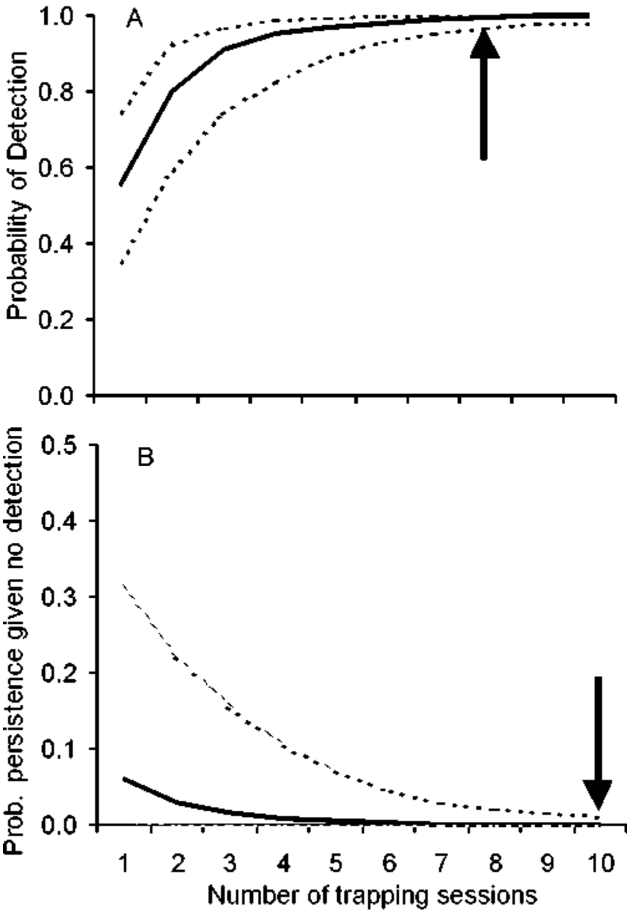

Fig. 2 Modelled probabilities of (A) detection and (B) persistence of stoats on Resolution Island out to 10 trapping sessions, using information obtained from stoats captured during the initial knockdown period (three trapping sessions). Dotted lines represent 95% credible intervals. Arrows indicate (A) a lower credible interval of 0.972, i.e. a probability of 0.028 that stoats present on the island would not be detected after 8 trapping sessions, and (B) an upper credible interval of 0.008, i.e. a 0.8% chance that stoats would remain on the island without being detected.

eastern part of the island (95% Credible Interval 72–140). This translated to a density of 1.6 stoats km^{-2}. The estimated probability of detection in traps increased very quickly with the number of sessions and was projected to have a lower 95% credible interval (conservative estimate) of 0.972 after eight trapping sessions (Fig. 2). Using these results to predict the probability of persistence in the confirmation phase (when stoats are no longer being detected in traps), we found that after ten trapping sessions with no stoat detections, the conservative upper 95% credible interval would be 0.008; a 0.8% chance that stoats remain on the island without being detected (Fig. 2).

DISCUSSION

Knowing the initial population size and detection probability of an invasive species is highly informative for eradication efforts. Furthermore, independent estimates using different methods are rarely obtained, so having multiple measures of these parameters increases confidence in the estimates. We were able to compare empirical estimates using non-invasive genetic sampling with data obtained from stoats captured during the initial knockdown phase of the eradication attempt on Resolution Island.

Non-invasive hair tubes identified about 25% of the stoats that were subsequently captured in kill-traps during the initial knock-down phase of the eradication attempt. Hair tubes were thus less effective detection devices relative to kill-traps once differences in their deployment are considered. Our low density estimates from hair tube sampling may have three origins. First, stoats may have been neophobic to hair tubes. Trap boxes had been in situ for several months prior to knock-down and had received two rounds of pre-baiting, so stoats may have become more used to their presence relative to the newly-placed hair tubes. Consequently, we could have sampled a smaller subset of the population. Second, in retrospect the period for hair tube sampling was insufficient to give precise density estimates. An interval of 2–3 weeks may have been more appropriate in order to make a direct comparison with the kill-trap data. Finally, the proportion of clean genotypes obtained from samples was only 60%, so the remaining samples, if resolved, would have increased the DNA-derived density estimates.

Our data suggest that kill-traps efficiently detected stoats at the moderately low density of 1.4 stoats km^{-2} measured on Resolution Island (see King and Murphy (2005) for other NZ stoat density estimates). We were also able to provide an informal, independent assessment of trapping success during the knockdown phase and conclude that it was >90%. This is particularly important for the current management of invasive mammals on Secretary and Resolution Islands, where traps are used in perpetuity to increase the chance of resident stoats being trapped and to prevent incursions from the mainland (Edge *et al*. 2011). Those naive stoats that do occasionally swim to the island (McMurtrie *et al*. 2011; Elliott *et al*. 2010) are likely to encounter an effective kill trap soon after arriving. However, it seems that some stoats survived the initial kill trapping and might represent trap-shy or narrow-ranging individuals. Female stoats usually retain between six and 13 blastocysts inside the uterus for up to a year (King and Murphy 2005), so survivors of a trapping programme will strongly contribute to the continuation of a stoat population in an area.

We were able to provide reasonably precise estimates of g0 and σ, which usefully tested the trap spacing on the island (McMurtrie *et al*. 2008). Our estimates of the spatial detection parameters are similar to other published studies (e.g., Smith *et al*. 2008; Efford *et al*. 2009), and gave an estimate of home range radius for stoats (c. 486m) similar to but slightly less than many of those derived by radio-tracking (King and Murphy 2005). So the initial goal of having a maximum of 700 m from any point on the island to the nearest kill-trap (McMurtrie *et al*. 2008) now seems to have over-estimated resident stoat home range sizes.

Catch-effort modelling of the data obtained from kill-trapping gave a less biased measure of the initial stoat population density prior to the knock-down, and was also useful for obtaining an independent estimate of the probability of detection for the current trap array. We could then predict how many trapping sessions would be required before being confident that eradication of stoats from Resolution Island had been achieved (assuming no in-situ breeding and no further incursions from the mainland). This knowledge is of little use at present, as stoats still inhabit the island (P. McMurtrie pers. comm., Feb 2011). A more useful analysis would be to model *in situ* breeding and likely immigration rates, which we are currently undertaking. The proposed Bayesian modelling approach will ultimately incorporate both the kill-trap and the genetic-mark-recapture data to provide improved estimates of the initial population size. The improved model will also incorporate the sex ratio bias, population growth rate, and the ongoing probability of immigration from the mainland. Improved modelling should also account for the possibility of decreasing detection probabilities as the population is reduced to near zero. Further, we have now established a genetic database of stoats from the island prior to the eradication, which can be used in the future to infer whether captured individuals are survivors or recent arrivals.

The attempted eradication of stoats from Resolution Island represents a large, complex and ambitious project. A key component of the planning and implementation of the eradication programme was to learn as much as possible about stoat behaviour and trappability on the island in order to adapt the operational aspects of the programme through time (Edge *et al*. 2011). We provide evidence that the kill-trap devices chosen, strong emphasis on pre-baiting to avoid neophobia, and ongoing use of the control tool (kill-traps) as a surveillance device were sound operational decisions for the eradication of stoats on Resolution Island. However, an increased density of kill-traps may be required if eradication is to be achieved. The DNA sequencing techniques we developed represent an important advance, but further research that reduces the problems of mixed samples would be beneficial. Finally, to ensure a successful programme, future work is needed to better understand detection probabilities at the very low population densities of stoats on Resolution Island and to combine multiple sources of uncertain, imprecise or sparse information.

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Animal welfare and ethical issues in island pest eradication

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Abstract Island pest eradications almost always involve killing, often of large numbers of animals. Future eradications must pay more attention to the issues this raises, not only because the issues are important in themselves but also because eradications sit within a wider context of increasing welfare and ethical concerns about animal pest management. The welfare issues include the pain and suffering caused directly by the control method used (to both target and non-target) and any flow on effects (eg. trophic cascades), while the ethical costs relate more to the justification and outcomes of the eradication programme. Eradication programmes will always have uncertainty related to funding, ability to target all individuals, and probability of reinvasion. This uncertainty means such programmes should only proceed and can only be defended on ethical grounds if they are structured in such a way that learning is maximised and applied to reducing uncertainties in future operations. Because structuring eradication programmes as learning experiments will have additional costs (eg. additional monitoring) this approach raises the issue of how much we are willing to pay for that learning and for more rapidly mitigating the welfare and ethical costs of island pest eradication. The consequences of not doing so may put at the risk the support for future eradication programmes.

Keywords: Non-target effects, harm, toxins, costs, benefits, learning experiments

INTRODUCTION

In some countries, the management of mammal pests is coming increasingly under the spotlight because of issues about the ethics of lethal control and the welfare impacts of the various pest control methods employed (Thiriet 2007; McEwen 2008; Warburton and Norton 2009). Animal welfare concerns may also at times be linked to other underlying political motives, such as the hunting lobby's interests in maintaining invasive species as game animals. Some welfare conflicts have been clear and public, such as the aerial 1080 poisoning of possums in New Zealand (Fisher *et al*. 2008), the culling of wild horses in Australia (Nimmo and Miller 2007), or the control of white-tailed deer in North America (Warren 1997). To date, however, the eradication of pests from islands has generally not been subject to the level of controversy that has attended mainland control operations, although there are notable exceptions (eg. Anacapa Island rat eradication; Howald *et al*. 2005). One reason for this is that island eradications have so far mostly involved uninhabited and often remote islands. This is changing as mammal pest eradication are proposed more often for islands that have permanent human habitation, pastoral uses and/or are close to highly populated mainland areas. The heightened visibility of such projects often gives rise to controversy. For example, it is doubtful if the removal of introduced hedgehogs from islands off the west coast of Scotland (Jackson 2003) would have engendered such debate if it had happened on one of the more remote, uninhabited islands in the southern Atlantic. Proximity to mainland populations brings closer scrutiny of environmental risks, such as water contamination and non-target impacts and also closer scrutiny of the core justifications for pest management and the tools it employs. Often public opposition is generated from being uninformed or through lack of information, so it is essential for the successful development and management of an eradication programme that public education is considered as important as technology and funding (Simberloff *et al*. 1997).

Three key prerequisites must be satisfied before eradication is likely to be achievable (Parkes 1993; Cromarty *et al*. 2002). These focus on animals mainly in relation to the need to kill them: all animals must be put at risk by the control tools; they must be killed faster than they can breed; and there must be no immigration. There is little explicit attention paid to the harm done to the animals during the eradication operation and to local cultural issues about the treatment of animals. We argue in this paper that future eradications must pay more attention to harm done not only because it is important in itself but also because pest eradications sit within a wider context of increasing welfare and ethical concerns relating to animal pest management, increasing animal welfare guidelines, and changes in laws and regulations (Meerburg *et al*. 2008; Warburton and Norton 2009; Yeates 2009).

ISSUES ASSOCIATED WITH ERADICATIONS

Failure

The number of attempts to eradicate mammal pests, particularly rodents, from islands (Table 1) has been increasing steadily (Nogales *et al*. 2004; Campbell and Donlan 2005; Clout and Russell 2006; Howald *et al*. 2007). This trend has been largely self-reinforcing, with success breeding success, leading to operations on increasingly larger and/or more remote islands, and attempts to eradicate multiple rather than single pest species (Parkes and Panetta 2009). The increase in eradication attempts has not, however, been accompanied by a drop in the failure rate, at least for rodents (Parkes and Panetta 2009). This implies that the absolute number of failed operations has increased, which is surprising given the much greater emphasis in recent times on feasibility studies and risk management, including general agreement about the criteria for attempting eradication (Parkes 1993; Bomford and O'Brien 1995; Cromarty *et al*. 2002; Parkes and Panetta 2009).

From a welfare perspective, failed eradications may have huge cost and little benefit, and so are of major

Table 1 Number and percentage of successes and failures of eradication attempts for various mammals (based on Nogales 2004; Campbell and Donlan 2005; Clout and Russell 2006; Howald *et al*. 2007). *= New Zealand data only.

Species	Successes	Failures
Rats (3 spp)	159	15 (8%)
Mice	30	7 (19%)
Goats	120	10 (8%)
Cats	79	17 (18%)
Rabbits*	17	2 (11%)

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concern. The largest islands on which eradications have failed for rats, cats and goats, for example, were 1815, 650,000, and 28,510 ha, respectively. Failed eradications may mean that tens to thousands of the target pest have been killed or harmed without achieving the goal of the operation. In the worst case, the failure to eradicate means that there is no further management of the pest species on the island and animals have died to no good purpose, or at best for a temporary reduction in their impacts.

Non-target impacts

Most, and probably all, eradications also harm and kill non-target species (eg. Cowan 1992). Minimising such harm is a major component of eradication planning, and the implicit assumption in most eradication feasibility studies is that the benefits of eradication outweigh the costs, including non-target species impacts. Such arguments do not always assuage public concerns as expressed, for example, after the recent deaths of gulls and bald eagles (*Haliaeetus leucocephalus*) during the rat eradication on Rat Island in the Aleutian Islands (see http://www.all-creatures.org/articles/ar-island.html accessed 27 January 2010). Single species pest eradications may also have perverse effects on non-target species, notably through trophic cascades and mesopredator release, and in extreme cases drive non-target species towards extinction (Courchamp *et al*. 1999; Roemer *et al*. 2002). In manipulating populations to the extent of eradication, we need to be sceptical about our motives and about our ability to intervene successfully in large, complex systems (Jamieson 1995).

Choice of eradication methods

A range of methods have been used to eradicate pests from islands (Table 2). All methods have welfare issues for the target and non-target animals involved. Different control tools have different welfare impacts (eg. Mason and Littin 2003), and this has given rise to research into the relative humaneness of control methods and decision support systems as aids to pest managers to assist them make informed choices about methods of control (eg. Littin *et al*. 2004; Fisher *et al*. 2008; Sharp and Saunders 2008).

Most island pest eradications involve rodents, and 70% of those have involved the use of non-selective toxins, particularly anticoagulants (Howald *et al*. 2007). The most commonly used of these, brodifacoum, is increasingly the focus of concerns. These derive from the potential environmental risks associated with bioaccumulation and persistence in carcases and sub-lethally poisoned animals and also the welfare impacts induced by its mode of action (Mason and Littin 2003; Paparella 2006; Meerburg *et al*. 2008). Brodifacoum use is now restricted for domestic use in the USA and the EU, and in New Zealand is no longer used by the Department of Conservation for mainland pest control. Other rodenticides, such as diphacinone, are being investigated as alternatives to brodifacoum for island eradications. The main driver for these assessments is not animal welfare but reducing the risk of secondary poisoning through use of compounds with significantly shorter tissue residue half-lives.

Table 2 Summary of methods used for eradications of mammal pests from islands

Methods	Welfare Issues
Trapping	
Cage traps	Stress, self-injury
Leg-hold traps	Stress, self-injury, trap injury
Kill traps	Time to unconsciousness, cause of death
Drowning traps	Stress, time to unconsciousness, cause of death
Poisoning	
Acute toxins	Symptoms, time to unconsciousness,
Anticoagulants	sub-lethal dosing, persistent effects (e.g., via residues), cause of death
Hunting	Wounding, stress
Biological control	Symptoms, time to unconsciousness
Judas animals	Reproductive manipulation, surgery

QUESTIONS ASSOCIATED WITH ETHICS AND WELFARE

What are the ethical and welfare issues in island pest eradications?

In simple terms, welfare issues include the pain and suffering caused directly by the control method used (to both target and non-target) and any flow on effects (eg. trophic cascades), whereas the ethical costs relate more to the justification and outcomes of the eradication programme. Ethical issues are more complex because they relate both to the concept of eradication itself and to the specific operation under consideration. A further issue is that of dealing with uncertainty. In such situations, the Precautionary Principle may be applied; namely, acting to avoid serious or irreversible potential harm, despite lack of scientific certainty as to the likelihood, magnitude, or causation of that harm.

Two criteria used to assess the feasibility of eradication (Parkes 1993; Bomford and O'Brien 1992, 1995; Cromarty *et al.* 2002) are particularly relevant to welfare and ethical issues. The first states that the benefits of eradication must outweigh the costs, which is a utilitarian view (Singer 1990). Although the benefits and costs are usually considered to be monetary, there is no reason why the same balancing of costs and benefits should not be undertaken from a welfare perspective. Such an approach underlies the application of various national codes of animal welfare (eg. http://www.biosecurity.govt.nz/ regs/ animal-welfare/ stds accessed 1 February 2010). Most island eradications have been based on the premise that the long-term benefits to the at-risk indigenous species outweigh any welfare impacts in the short-term. Nevertheless, any welfare impacts in the short term should be minimised by explicit consideration of animal welfare as a criterion when selecting eradication methods, and preferably by selecting those methods that pose the least harm. The second states that the techniques used for eradication must be acceptable to stakeholders and communities. In effect, this is usually another cost-benefit decision by those involved, balancing the need for efficient and effective killing methods to minimise risk of failure against the various community views on the ethical issues involved and welfare costs of the methods used.

How might the welfare and ethical issues be addressed?

Welfare impacts can be described by a formula that accounts for the direct impacts on target and non-target species, and includes flow on effects as part of the non-target impacts.

$$WC_{Total}=(WC_{TL}*N)+(WC_{TSL}*N)+(WC_{NTL}*N)+(WC_{NTSL}*N)$$

where:

WC_{TL} = Welfare cost to target species that are killed

WC_{TSL} = Welfare cost to target species that are sub-lethally poisoned or injured

WC_{NTL} = Welfare cost to non-target species that are killed

WC_{NTSL} = Welfare cost to non-targets that are sub-lethally poisoned or injured.

N = the number of animals in each of these categories

The welfare costs are minimised when N is minimised and the method(s) chosen has the least welfare cost. Eradication programmes should therefore aim for success as quickly as possible to minimise any births during the process, and use the most humane and target-specific methods. If the indigenous species that is threatened by the invasive has very high conservation value (eg. is the world's sole population), the benefits of eradication are likely to be considered greater than if the indigenous species also occurs elsewhere. Benefits and costs thus need to be weighed against each other, and a higher cost (including welfare costs and uncertainty) might be accepted when the benefits are exceptionally high.

Ethical issues can be addressed by considering a series of questions. Based on the principles underlying the ethical approval of the use of animal in research, Yeates (2009) presents a generic ethical decision-making algorithm to assist this process for pest management (Fig. 1).

In the case of island pest eradication, two sets of questions should be asked. First, do the conservation benefits actually justify the killing of the exotic species? The justifications for island pest eradications have encompassed a wide range of projected benefits – how should different benefits be contrasted and/or combined? The number of island pest eradications is increasing but, at a global level until recently, little thought appears to have been given to prioritisation – on how many and which islands is it crucial to remove invasive alien species? Second, is the risk of failure too high? Will perverse outcomes result in minimal benefits, will the eradication fail because of cessation of funding or because of unforeseen technical problems, or will the benefits of successful eradication be lost if the islands cannot be secured from natural or human-assisted future invasions?

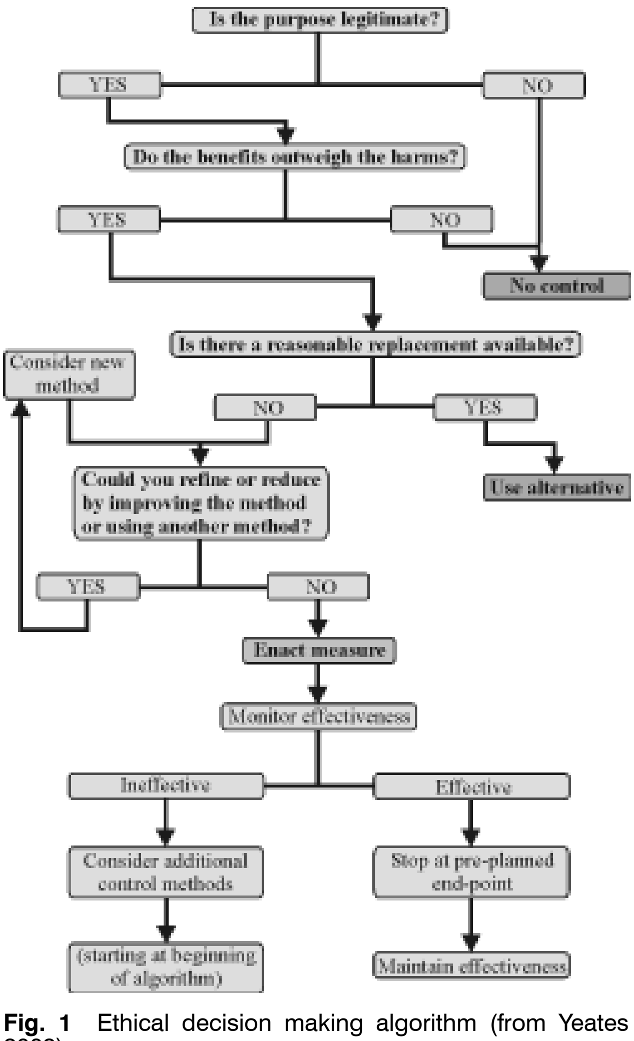

Fig. 1 Ethical decision making algorithm (from Yeates 2009).

These issues all contribute uncertainty to eradication attempts. Failures highlight the welfare and ethical issues, and justifiably raise the bar for future attempts. To address this, programmes must identify uncertainties at the planning stage and develop mitigation strategies, which are done increasingly as part of eradication feasibility studies. Such approaches to reducing the risk of failure should be complemented by a learning-based strategy. This is the central feature of the ethic proposed by Warburton and Norton (2009): to ensure that even if eradication fails it provides knowledge to improve future attempts. They suggest that this ethic can be made functional within an adaptive management framework that has as its first tenet the need to learn and reduce uncertainty (Walters and Holling 1990).

CONCLUSIONS

Welfare impacts (i.e. inhumaneness) of the eradication methods used are an increasing focus of opponents of the killing of invasive species, and some methods, such as anticoagulant toxins, will most probably continue to be a concern. However, even if eradication methods were 'humane', wildlife managers planning or conducting eradications still face ethical challenges. These revolve around whether the number of animals killed is justified in terms of the conservation benefits achieved, especially when the uncertainty surrounding an eradication attempt is high, with the risk that many animals may be killed for no benefit. Eradication programmes will always have uncertainty such as that related to funding, ability to target all individuals, and probability of reinvasion. We propose that programmes with such uncertainty should only proceed and can only be defended on ethical grounds if they are structured in such a way that learning is maximised and applied to reducing uncertainties in future operations. Because structuring eradication programmes as learning experiments will have additional costs (especially for additional monitoring) this approach raises the issue of how much we are willing to pay for that learning and for ensuring the welfare and ethical costs of eradication programmes are reduced.

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Removal of red deer (*Cervus elaphus*) from Anchor and Secretary Islands, Fiordland, New Zealand

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Abstract Red deer (*Cervus elaphus*) have been successfully removed from Anchor Island (1130 ha), progress has been made toward removing them from Secretary Island (8100 ha), and a start has been made against the population on Resolution Island (21,000 ha) in Fiordland, New Zealand. The programme on Anchor Island ran from July 2002 until December 2007, and removed 29 deer. Team hunting combined with the use of tracking dogs was the key method. The Secretary Island programme began in 2006 and is continuing. This programme was planned in three phases. The first was to reduce the population within 2 years, and 542 deer (estimated to C. 80% of the population) were killed mostly by ground-based hunting with some helicopter hunting. The second was to remove the survivors, and 91 deer had been removed up to the end of June 2010, mostly by helicopter hunting. The final phase will attempt to detect and remove any immigrant deer. DNA extracted from hair samples was used in an attempt to calculate an initial population size, but the deer were too closely related for this method to be useful. However, the high level of relatedness indicates that there have been few female immigrants to Secretary Island, suggesting reinvasion may be a rare event. The removal of deer from Resolution Island began in November 2009.

Keywords: Helicopter hunting, eradication, ground hunting, indicator dogs

INTRODUCTION

Islands are frequently used as sanctuaries for threatened species. There are approximately 90 islands over 1 ha in size within the Fiordland region of New Zealand, but almost all are sufficiently close to the mainland to be within swimming distances of stoats (*Mustela erminea*) and red deer (*Cervus elaphus scoticus*). Until relatively recently, these islands were not considered likely to provide safe sanctuaries for threatened species, but over the last ten years several pest populations have been eradicated from Fiordland's islands allowing native species to be translocated to them (Elliott *et al*. 2010). Although this ongoing programme focuses primarily on invasive predators, there is an increasing focus on removing all feasibly eradicable pests. For red deer, the long-running programme of sustained deer control in the Murchison Mountains of Fiordland, to protect the habitat of endangered takahe (*Porphyrio hochstetteri*), has helped develop and refine methods suited to the Fiordland environment (Fraser and Nugent 2003) and provided hope that removal of all deer from Fiordland's islands might be possible.

Here we document one completed (Anchor Island; 1130 ha) and one current (Secretary Island; 8100 ha) programme to remove red deer from progressively larger islands in Fiordland. We also note the start of a third attempt on Resolution Island (21,000ha) (Fig. 1).

MAIN FINDINGS

Study Area

The coastal Fiordland region has a wet, cool, temperate climate. Westerly or north-westerly winds bring most of the rainfall while southerly or south-westerly winds can bring snow to the higher altitudes, particularly in the winter months. Anchor and Secretary Islands (Fig. 1) lie at the entrance to glacial fiords and have been heavily modified by glacial erosion.

Anchor Island rises steeply on the eastern side to a high point of 417m, with low hills to the west. The highest ridges are capped with small areas of tussock and shrubland, while the lower ridges and hill slopes are covered with a mixed podocarp-broadleaved-beech forest, with coastal scrub, especially on the western shores. Secretary Island is much larger, steeper and more rugged. It has a greater diversity of forest and shrubland cover, with many open landslips. The island rises steeply to over 1100 m and has several areas of alpine vegetation.

Anchor Island Campaign (2002-2007)

As in many parts of Fiordland, deer densities on Anchor Island were probably quite high before the 1970s, but were considerably reduced by commercial hunting during the 1970s and 1980s. Hunting by fishermen and recreational hunters probably helped keep numbers low after that. During an initial survey in 2001, the survey team noted well worn deer trails and estimated that about 20 deer were present, mostly on the western side.

Although hunting, as part of the eradication campaign, began July 2002, the control plan was not formalised until October 2003. That plan acknowledged the potential for deer to re-invade the island and a need for ongoing surveillance and control (M. Mawhinney pers. comm.).

Use of poisoned baits was not favoured because deer food was relatively abundant on the island. A programme based initially on the use of ground hunters and dogs was therefore developed. The waters around Anchor Island are generally sheltered, with landing possible on most shores, so hunting operations were based from a boat. Most hunting involved week-long trips by a team of 5-10 hunters, with a total of 24 trips being completed over the period 2002 to 2007. Although 34 different hunters were involved, most hunting was conducted by just three of them. Initially, hunters worked independently, but after two years there was a switch to team hunting. By then, areas favoured by the remaining deer and previously used escape routes had been identified. Hunters were placed at strategic spots to ambush escaping deer that were being hunted by others in the team. Communication by hand-held radio was crucial in repositioning hunters when hunted deer changed course or avoided ambush. Several smaller islands between Anchor Island and the nearest other deer populations were checked regularly for deer that might use these as "stepping-stones" to invade Anchor Island, or to escape from it.

Helicopter-based hunting was used on several occasions. However, deer made little use of the larger open areas so helicopter hunting was ineffective, although the second last deer killed was shot from a helicopter in the highly favoured western coastal scrub/forest habitat.

Ten self-attaching snare collars (Taylor 1969) that incorporated a radio transmitter were set from December 2003, in an attempt to collar deer so they could be radio-tracked and shot. Deer did pass through and knocked down some radio-collar sets, and some interference by seals was also recorded. Two collars disappeared from where they were set: one was recovered C. 250 m away with the clip

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not fastened, while the other was found 800 m away with the clip fastened but on the ground by an antler-thrashed tree, suggesting it had become tangled in a stag's antlers but later came free.

From 2004, barrier fences were used in an attempt to confine deer to trails (and to provide improved sites for setting self-attaching snare collars). Initially, portable electric fence tape was used as a visual guide. This changed to the use of posts and wire netting to establish permanent barrier fences on key trails, where topographical features already channelled deer movement. Gateways were also established as sites for setting self-attaching collars or placing ambush hunters. Deer were recorded using the gateways of all five barrier fences late in the programme, and a barrier fence was pivotal in the shooting of the last deer.

On a few occasions, nets were suspended across deer trails or barrier fence gateways in the hope that deer fleeing the hunting team would be caught in them, but none were caught this way.

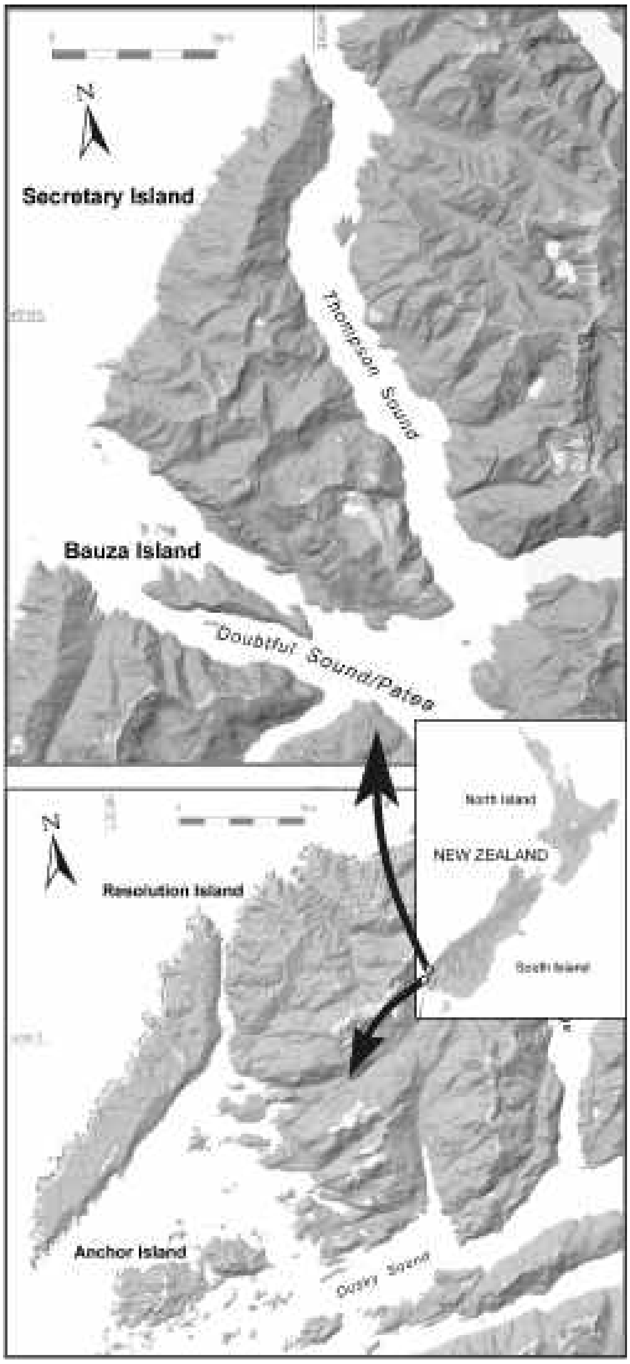

Fig. 1 Location of Secretary, Anchor and Resolution Islands.

In total, 29 deer were removed from Anchor Island and two neighbouring small islands between July 2002 and December 2007 (Table 1). Checks of the island in December 2008 and December 2009, where hunters (with dogs) familiar with the island checked all areas of known preferred habitat, found no deer.

Table 1 Numbers of deer shot (by method) and effort; Anchor Island 2002-2007.

	Kills by Hunting Method			Effort	Hours/
	Individual	Team	Helicopter	(hrs)	Deer
2002	8	0	0	324	41
2003	5	0	0	774	155
2004	3	2	0	824	165
2005	0	9	1	1244	124
2006-07	0	1	0	456	456
Total	16	12	1		

Secretary Island Campaign (2006-ongoing)

In 2006, a second campaign began on Secretary Island. Mark and Baylis (1975) had confirmed the presence of a small resident population of red deer there in 1970, prompting an effort between 1970 and 1989 to remove the deer and prevent re-invasion. That effort was neither intensive nor widespread enough to eliminate the population (Brown 2005). In contrast, the new extirpation programme begun in November 2006 was better planned, with adequate resourcing of an intensive control effort that delivered comprehensive coverage of the island. A four year programme of work aimed to completely remove deer within that time, but annual reviews of progress were scheduled to allow regular reassessment of the feasibility of the goals and objectives.

As with Anchor Island, the Secretary Island campaign was conducted in conjunction with a stoat eradication campaign (see McMurtrie *et al*. 2011). The greater scale, complexity and difficulty of the Secretary Island campaign required a different mix of techniques to those used on Anchor Island.

As a first step, a hut and track network and radio repeater were established in 2005, prior to the stoat eradication programme (see McMurtrie *et al*. 2011). At the same time, an analysis of the likely issues and potential methods were presented in a scoping document (Brown 2005). Based on responses to that, an operational plan was developed in 2006 (Crouchley *et al.* 2007). Because reinvasion from the mainland was considered likely, the goal was the complete removal of the resident deer population (which we now term 'extirpation'; see Edge *et al.* 2011) followed by an ongoing monitoring and control programme to manage re-invasion. Because Secretary Island was larger than any other forested island from which deer had previously been eradicated, the programme was necessarily experimental in nature, with a key subsidiary aim being the development or refinement of methods for the proposed eradication of deer from the almost three-times larger Resolution Island.

For Secretary Island, three phases were planned, as follows:

1. A 'knock-down' phase, aimed at reducing the population by 80% within two years
2. A 'mop-up' phase, aimed at removing all surviving residents in Years 3-4
3. A 'maintenance' phase, aimed at detecting and removing new arrivals in perpetuity

The plan included reviews at the end of each phase, as there was some expectation that objectives would need to be revised or mop-up work extended beyond Year Four.

Ground and helicopter hunting were the main methods used to achieve initial 'knock down'. Additional methods such as capture pens, baits and lures, self-attaching transmitter collars and Judas animals have been introduced during the mop-up phase.

Ground Hunting: Priority was given to contracted ground hunters during 'knock-down' so that observations and data collected by the hunters could be used to assess progress and plan tactics. These hunters recorded the GPS locations of any deer shot or seen, and of any fresh deer sign. They used rifles and indicator dogs and worked separately from huts located near the centre of nine ground hunting 'blocks', usually for 4-5 days per block. They were rotated around the blocks to ensure that each hunter gained a good knowledge of the whole island and that each block was hunted by multiple hunters. The programme was designed to pulse hunting effort, with individual blocks being rested, to allow deer to return to preferred areas, for at least two weeks between each hunting session. Blocks on the western side of the island were known to hold the highest densities of deer so were hunted more often.

Unlike Anchor Island, team hunting was not often practical because of the more rugged terrain and larger size of the areas, but this technique may still be used during 'mop-up'.

Helicopter Hunting: Helicopter hunting is very effective in open unforested areas, so the alpine grasslands, scrub areas, numerous slips, open coastal fringe, and areas of open canopy forest on Secretary Island were more suited to this method than on Anchor Island.

Four different pilots and three models of helicopter were used (Hughes 500, Robinson R44 and R22) to vary the hunting style and helicopter attributes (such as noise level) that might influence effectiveness. All pilots were experienced aerial hunters. Two shooters were used with the larger machines (Hughes 500 and Robinson R44) with a shotgun being used from the front door and a high-powered rifle from the rear door. The smaller R22 helicopter, with a single shooter using high-power rifle, was utilised more during Years 3 and 4 when fewer animals were being shot. Helicopter hunting was carried out periodically throughout each year as weather conditions permitted, with at least two weeks between hunts, and a total of 9-15 hunts per year. The GPS location of each deer seen or shot was recorded, plus track logs for hunting flights.

Barrier Fences: While barrier fences might enhance the effectiveness of ambush team hunting, few sites suited to fencing were found, and only one fence was built (in conjunction with a capture pen).

Capture Pens: A total of 17 capture pens were built in the first two years, using 150 x 1900 mm wire mesh netting with wooden posts for gateways and corners, and steel standards between. Each had two drop-down gates, with a thin copper trip wire. Limited areas of flat terrain meant that the finished pens are generally small (100-200 m^2). Each pen contained a water container and some natural food. Pens were remotely monitored using a VHF radio repeater to deliver a number code via e-mail. The status of the gates was monitored and there was an additional trip wire on the pen fence.

Self-attaching Radio Collars: A self-sizing design, different to that used on Anchor Island, using a sliding loop that moves over a series of 'barbs' (Kirchoff and White 2002) was used for seven collars set during Year 2. Several different methods of setting were used. Some deer passed through these sets but none were collared. Work is continuing on design and setting techniques.

Judas Deer: After 'knock-down', the use of the 'Judas' technique was explored. This method uses radio-collared animals to guide hunters and enable them to find and kill the uncollared animals they associate with. To enhance the 'finding' power of the Judas deer, the concept of prolonging oestrus was investigated. This method has been applied to Judas goats to help eradicate goats from several Galapagos islands (Campbell *et al.* 2007). Two hinds were captured from the nearby Murchison Mountains and held in captivity for a period to be sterilised (through tubal ligation), fitted with a 400-day hormone implant, and tagged prior to release on Secretary Island in April 2009. This process was too late to be useful in the autumn 2009 rut, but should be effective during the 2010 rut. In addition, in 2009, a stag from the Murchison Mountains was fitted with a satellite linked GPS tracking collar and released on the island. The objective was to monitor his activity in relation to open habitats over the 2009/10 summer and 2010 autumn and to assist in planning the timing of helicopter hunting. Data collected on favoured sites will identify sites to check for remnant animals on the island. These data should be of particular value during the rut period when this animal will be searching for hinds.

Other Methods: A variety of food, scent and salt lures were trialed on wild red deer in accessible mainland forest sites. These trials did not identify any lure or bait that would attract wild deer any better than natural food baits. Deer did show interest in some of the baits and there was some indication that time of year or seasonal factors may have had some influence in bait attractiveness.

Trail monitoring cameras were trialed in conjunction with bait and lure trials. From Year Two, a small number of cameras were used on the island to help identify the presence of deer at key hunting or capture pen sites, and they provided some useful information on the presence (or absence) of deer at several sites of interest, especially during the 'mop-up' phase.

FLlR (Forward Looking Infra-Red) equipment has not been used because field inspection identified limited potential for its useful application in this project (P. McClelland pers. comm.), but may be re considered for use later in the programme.

Monitoring: An attempt was made to estimate the pre-campaign population size, using a DNA-genotyping mark-recapture approach in which fawns are effectively used as recaptures of their parents (Crouchley *et al*. 2007). The technique had been developed and successfully pilot tested on red deer in the Murchison Mountains in eastern Fiordland (Nugent *et. al*. 2005). Hair samples were collected from all ground-shot animals, and in Year 1 useful genotypes were successfully obtained for each of 72 adult female and 13 fawns. However there was very little variation within the standard panel of 14 genetic markers used. As a result there were a number of instances in which the set of alleles for a fawn matched those from two of more adult females (i.e. some fawns could have been assigned to more than one potential mother). That rendered the approach invalid. The solution of analysing a much larger set of markers was not explored because of cost. However, an important upside is that the data show only a low level of genetic diversity on Secretary Island population, with many of the rarer alleles found in mainland deer not detected. A key implication is that the founding stock for the population must have been small, indicating that the reinvasion (immigration) rate is likely to be low.

Overall results: In the 'knock-down' phase, 67 hours of helicopter hunting and 664 days of ground hunting resulted in 542 deer kills. In the first year, almost two thirds were helicopter kills, but since then the balance has slightly favoured ground hunting (Fig. 2). However, the cost per deer kill has been substantially lower for helicopter hunting, and in the second year (for example) was only one third of that for ground hunting.

Kills per unit hunting effort have generally declined over the campaign, but changes in hunter skill, experience and hunting method (and also in the wariness of the remaining deer) mean that the data do not provide a robust estimate of changing deer abundance. However, the lower kills rates and a subjective assessment of the abundance of deer sign indicated a major reduction in deer density. By the end of the 'knock-down' phase the ground hunters were finding little or no sign of deer in previously favoured locations. It was estimated that less than 100 deer remained. A similarly subjective estimate of 20 deer on Anchor Island was approximately correct, when the total killed (29) is adjusted down for deer born during the campaign.

Of the 542 deer shot during 'knock-down', only 461 were present at the start of the campaign, the remainder having been born since 2006. If the estimate of 100 deer remaining at the end of that phase (including some born after campaign start) is accepted, the maximum size of the initial population was ~560, suggesting that the desired 80% reduction had been achieved.

The two year 'mop-up' phase was completed on 30 June 2010. A total of 91 deer were killed (57 helicopter, 32 ground hunting, and two captured in pens). Complete removal of the red deer population was not achieved by this date but field observations, records of deer sign suggested that fewer than 25 animals were likely to remain.

DISCUSSION

Success on Anchor Island, coupled with success in managing other pests on other smaller islands where there is potential for pest animals to re-invade and the increased experience with managing successful deer control programmes at mainland sites, provided managers with the confidence to plan increasingly larger pest eradications and to cost those realistically.

The Anchor Island campaign demonstrated how continued refinement of methods and targeting effort based on experience could greatly increase efficiency, to the extent that the hunting effort per deer kill was lower in Year Four (when tactics changed to team hunting and the hunting effort targeted individual animals) than in Years Two and Three. Likewise, the use of hunting success rate data to target ground hunter effort at areas with highest kills rates (and presumably therefore the most deer) enabled hunters to maintain the same annual kill rates despite reducing deer numbers (Fraser and Nugent 2003). This approach of targeting effort and continuously modifying tactics is likely to be crucial in removing all deer from Secretary Island. In addition, the use of technological developments such as trail monitoring cameras, DNA monitoring, FLIR, and tracking collars will help to improve our knowledge of animal behaviour and how to target the last few deer. Thus, while deer have so far been extirpated only from Anchor Island, the success and progress as planned on Secretary Island, provides some confidence that all programmes will ultimately be successful.

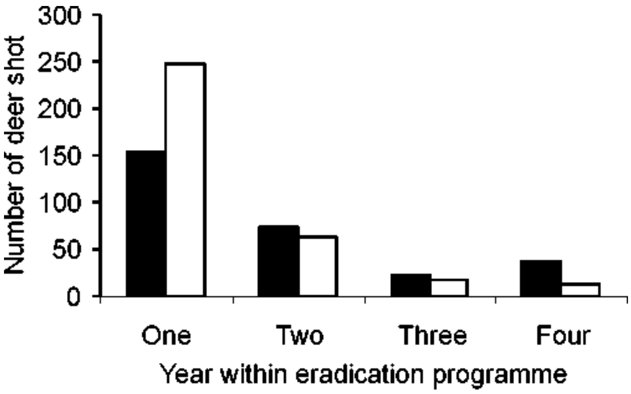

Fig. 2 Number of deer shot during the knock-down (years 1 – 2) and mop-up phases (years 3 – 4) of the Secretary Island deer eradication campaign (Filled bars = helicopter kills, open bars = ground hunter kills).

Given that all three islands are within swimming distance of mainland deer populations, the ability to detect and deal with any immigrants is essential. It will be possible to use the results of the current extirpation campaigns to interpret future surveillance surveys that find no deer (e.g., Ramsey *et al.* 2009), but the question of how often such surveys should be conducted once the resident deer are removed remains unknown and depends in part on the frequency of invasion events. While the low genetic diversity of deer on Secretary Island precluded the planned use of genetic mark-recapture methods for density estimation, it indicated that few deer ever invaded this island. As these few invasions are most likely to have occurred in the 1960s when deer numbers on the mainland were much higher, it now appears that the likelihood of deer re-establishing on Secretary Island is very low. DNA genotyping is likely to be useful in identifying where any deer killed after the 'mop-up' phase is a survivor or a new invader.

ACKNOWLEDGEMENTS

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DNA profiling – a management tool for rat eradication

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Abstract DNA profiling is a powerful tool for eradication planning and post-eradication management. We give an introduction to DNA methods for conservation, intended to be accessible to non-specialists with no previous knowledge of genetics. We illustrate the methods with a case study from Aotea/Great Barrier Island, New Zealand, where DNA methods have been used to manage eradications of ship rats (*Rattus rattus*). In initial management planning, DNA profiling gives evidence about reinvasion risk if an eradication is attempted. In the case of Great Barrier Island, we find that cliffs may be significant factors affecting reinvasion risk. After an eradication is attempted, DNA testing can determine whether new rats that appear are survivors of the eradication or reinvaders from another location, and can often determine precisely where the reinvaders have come from. This helps to focus management efforts for the prevention of future reinvasions.

Keywords: Eradication, genetic boundary, island, microsatellite, *Rattus*, reinvasion

INTRODUCTION

The creation and maintenance of island sanctuaries free of rodents is a major conservation focus in New Zealand. The most problematic invasive rodents include Norway rats (*Rattus norvegicus*) and ship rats (*R. rattus*), both of which can swim hundreds of metres, or hitch-hike to islands on boats. As rat eradication attempts become more widespread and more ambitious, we need to advance our understanding of reinvasion processes, including the swimming capabilities and tendencies of rats, and the frequency of accidental boat transport.

DNA profiling of rat populations is a relatively new tool for eradication managers. Several studies attest to its usefulness for managing rat populations on islands (Robertson and Gemmell 2004; Abdelkrim *et al.* 2007; Russell *et al.* 2010) and the mainland (Abdelkrim *et al.* 2010). DNA profiling can inform island managers in two ways. Firstly, it can uncover patterns of swimming in existing rat populations, by assessing the level of gene flow between different islands. Some islands are genetically isolated from each other, suggesting that either there is little migration between them, or there are social factors that inhibit breeding after migration. Other islands are genetically linked, suggesting high migration and interbreeding. We can study features associated with isolated or linked populations, such as the size of the water crossing, presence of cliffs, and accessibility of landing points. An understanding of features associated with high or low gene flow can help to suggest candidate islands for eradication in the future.

Secondly, DNA profiling can determine whether rats found after an eradication attempt are survivors of the eradication, or reinvaders from another source. This is vital for targeting the management response, either for improving biosecurity in the case of reinvaders or for examining eradication protocols in the case of survivors. Both outcomes can enhance our understanding for the future as well as for a specific situation. Reinvaders help to calibrate how genetic isolation translates to actual reinvasion rate. Survivors clarify our expectations about the short term effectiveness of a poison drop, especially among eradications that are eventually deemed 'successful' after the standard two-year follow-up period.

Our aim in this paper is to provide an accessible introduction to DNA profiling as a tool for eradication management, assuming no previous knowledge of genetics. Interpretation of DNA evidence is not always precise, and there is an immense and bewildering array of statistical analysis methods and software packages. Instead of aiming to be comprehensive, we will deliberately restrict our coverage to two genetic concepts, and attempt to explain these in enough detail for non-specialists to appreciate their power and limitations. The first concept is 'genetic distance' between populations, to measure genetic isolation of different islands, and we explore this using the distance measure F_{ST}. The second concept is of 'individual belongingness', which measures how well a single rat fits into each of several candidate populations, for example whether it is a survivor or a reinvader. For this we will describe the idea of genotype probabilities.

Our account is based on a study of ship rats in the archipelago surrounding Aotea/Great Barrier Island, New Zealand (28,500 ha; Fig. 1). We report results from extensive DNA sampling from 2005 - 2008, and link genetic structure to features such as cliffs and water distances. In 2008, an ambitious eradication focused on Kaikoura Island (530 ha) in the west, and we report the contributions of the DNA work to sourcing post-eradication rats that appeared

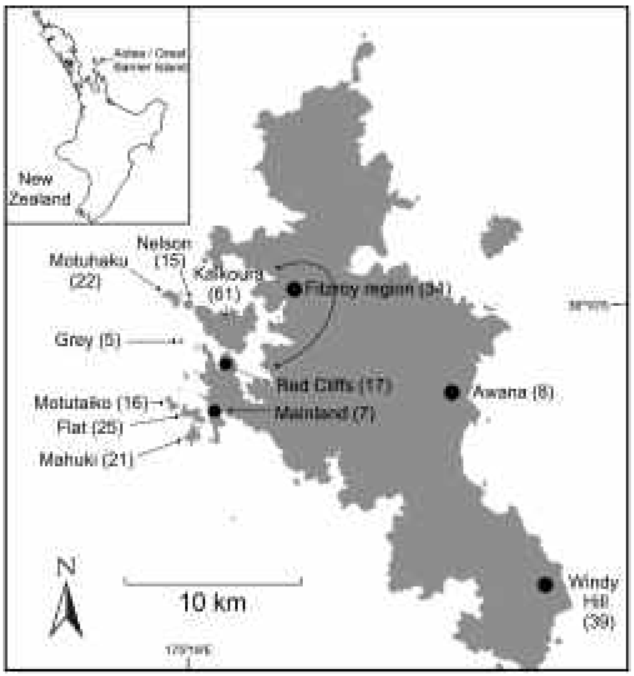

Fig. 1 Sampling locations on Great Barrier Island and surrounding islands. Numbers in brackets give the number of rats from each location for which DNA samples were submitted for genotyping. The Broken Islands are the group of Motutaiko, Flat, and Mahuki.

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on Kaikoura from early 2009 onwards. In 2009, a further eradication took place on the Broken Islands 3 km south of Kaikoura. We consider how DNA evidence could contribute to ongoing management of this region.

METHODS

Sampling

The Aotea/Great Barrier Island archipelago includes three island clusters: the Kaikoura chain comprising Kaikoura, Nelson, and Motuhaku; the Grey group of about 6 small islands; and the Broken Islands comprising Motutaiko, Rangiahua/Flat Island, Papakuri, Big Mahuki and Little Mahuki. From 2005 to 2008, we sampled a total of 270 rats from 12 locations (Fig. 1). We focused on the three island clusters, adjacent locations on the main island (Aotea), and two outgroups at Windy Hill and Awana. Rats were caught with snap-traps, and DNA samples corresponding to tail clips of about 4cm were preserved in 70% ethanol.

Eradications

In mid-2008, the Motu Kaikoura Trust began the eradication of rats from Kaikoura Island and nearby islands (Grey Group, Nelson and Motuhaku) using brodifacoum cereal baits spread by helicopter twice over two weeks in August and September. It was followed up with an intensive ground-based detection and response system on Kaikoura and the nearest parts of Aotea within swimming distance by rats. New rats were detected on Kaikoura by early 2009.

In June 2009, the Auckland Regional Council initiated rat eradication from the Broken Islands following a protocol similar to the protocol on Kaikoura. As of January 2010, no new rats had been reported from these islands.

Genetic loci and DNA profiling

A genetic *locus* (plural *loci*) is a position on a rat's DNA. For the type of loci that we use, every rat has two *alleles* at every locus, one inherited from each parent. The two alleles may be the same as each other, or different. When the rat reproduces, one of its two alleles is selected at random to be passed on to its offspring.

Some genetic loci contain molecular code for a specific physical trait, such as hair colour, in which case the outcome of this trait for a given rat will be determined by which alleles it possesses. However, many loci contain 'junk' or non-coding DNA known as microsatellites. These loci surround the useful loci like packaging in a box. They follow the same rules of genetic inheritance, but do not correspond to any physical trait, so they are prone to harmless coding errors or mutations. Over millennia, mutations create numerous available alleles for these loci, none of which do anything. The resulting genetic variety means that different populations can have very different genetic profiles at junk loci, so these are the loci chosen for DNA profiling studies and forensics.

The key to DNA profiling is the different proportions of alleles in different populations. On one island, 80% of the alleles at a junk locus might be of type A and 20% of type B, whereas on a neighbouring island, there might be 70% of type B and 30% of type C. If an unknown rat has an allele of type A, it must be from the first island, while if it has type C. it must be from the second island. Alleles of type B could be from either island but are more common on the second island, so we operate on the balance of probabilities. A conclusive decision requires not one but several loci, each of which sways the balance of probabilities one way or the other. The combined strength of about ten loci is often enough for a conclusive decision. This is the principle underlying *genetic assignment tests* – the process of assigning an unknown individual to a population.

Our study used ten microsatellite loci: D10Rat20, D11Mgh5, D15Rat77, D16Rat81, D18Rat96, D19Mit2, D20Rat46, D2Rat234, D5Rat83, D7Rat13 (Jacob *et al.* 1995). Details of the DNA extraction and amplification are given in Russell *et al.* (2010).

Can we tell the populations apart? Genetic distance

Some populations are more genetically distinguishable than others. For example, it is easier to distinguish between Asian and European populations of humans than between Scottish and English populations. The genetic differentiation depends upon the length of time since the populations split, their size, and the amount of ongoing migration between them. Similarly, the junk DNA in rats on an island can quickly develop allele profiles that differ from other islands, especially if the founding populations involved a small number of individuals. Substantial ongoing migration between island populations will keep them genetically similar.

The degree to which different populations are genetically distinguishable can be measured by a *genetic distance*. A widely accepted distance measure is F_{ST} (Wright 1978), where F denotes 'Fixation index', and 'ST' denotes 'Subpopulation within the Total population'. The 'subpopulations' can be seen as different islands and the 'total population' as the combined subpopulations.

To illustrate F_{ST}, we can think of two islands with the same numbers of rats, and a single locus with two possible alleles, A and B. If all A and B alleles from rats on both islands were put together and one allele drawn at random, the selected allele would vary between A or B. F_{ST} is the proportion of this variance that is explained by the differences in allele frequencies between islands. For example, suppose the two islands are identical, each with 50% allele A. Knowing which island a selected allele comes from gives no information about which allele it is, so the genetic distance between islands is $F_{ST} = 0$. However, suppose allele A is possessed by no rats on island 1 but all rats on island 2. The combined proportion of A from both islands is still 50%, but in this case, knowing which island the selected allele is from specifies exactly which allele it is. The island differences therefore explain 100% of the variance in allele selection, so their genetic distance is $F_{ST} = 1$. The same idea of partitioning variance can be extended to calculate F_{ST} when there are different population sizes, multiple alleles, and multiple loci.

In summary, F_{ST} measures genetic distance on a scale from 0 to 1. At 0 the populations are genetically indistinguishable and at 1 they are completely distinguishable, i.e. they are fixed for different alleles. A useful rule of thumb is that F_{ST} values from 0 to 0.05 denote little genetic distance; 0.05 to 0.15 denote moderate distance; and 0.15 and above signal large genetic distance and easily distinguishable populations (Wright 1978).

In this study, we calculated F_{ST} for pairs of adjacent populations using Genepop on the Web (Rousset 2008), available free from *http://genepop.curtin.edu.au/*. We further used the software FreeNA (Chapuis and Estoup 2007, *http://www.ensam.inra.fr/URLB/*) to correct our F_{ST} estimates for the possible presence of null alleles, which are alleles that do not show up on the DNA profile of an individual due to a mutation just outside the microsatellite region. Although we use the corrected estimates here, there is negligible difference between these and the estimates gained from Genepop. FreeNA uses the method of Weir (1996) to calculate F_{ST} after possible null alleles have been excluded.

Where does a rat come from? Genetic 'belongingness'

F_{ST} is used for measuring the genetic distance between two populations, such as two islands. We also need a method for measuring how well an individual rat fits into a given population, which is useful for two reasons. Firstly, if a rat has unknown origin – for example it is a reinvader to an island that has previously been eradicated – we can examine its fit to all possible source populations to estimate where it came from. This process is *genetic assignment*. Secondly, we can routinely examine the fit of all rats to all populations, which can reveal individual anomalies such as rats caught on one island that have the genetic characteristics of a different island. Such anomalies provide direct evidence of migration by swimming or by boat, and they cannot be detected by population-level distances such as F_{ST}.

To understand how an individual measure of 'genetic belongingness' works, we will use another example from human populations. If a blond man is seen walking down the street in Zanzibar, we might want to know where he comes from. Blondness is common in Sweden – perhaps 80% of Swedes are blond – but people are also blond in many other countries. If we give the man an 80% belongingness probability for Sweden, it means that 80% of Swedes are blond, not that the man is 80% likely to be Swedish. Unfortunately, there is no way of calculating the man's probability of being Swedish, much as we would like to. We can only say how common his blond characteristic is in Sweden, and compare with how common it is in other countries.

This idea is a common source of confusion in genetic reporting. If we replace the human analogy with rats on islands, we can change our 'blond man' to a 'rat with its observed set of alleles', and replace Sweden by a possible island source for the rat. All we can say about the rat is that it is more or less typical of different islands, just as blond men are common in Sweden but less common in Italy. We cannot say that the rat is 80% likely to come from any island, just as it is absurd to suggest that 80% of blond men in Zanzibar are automatically forced to be Swedish.

To help to keep the distinction clear, we will refer to the probabilities as measures of *genetic 'belongingness'* or *'fit'*. A blond man looks as if he *belongs* or *fits in* to Sweden, but this does not exclude him from fitting equally well or better to Denmark or elsewhere. Similarly, given a specific rat, we calculate the probability of this rat's alleles in each of our potential islands. Because every rat is unique, these probabilities will usually be very small, so we take logs to convert tiny numbers back to a manageable scale.

The measure of 'belongingness' or genetic fit that we use is called the *log genotype probability.* The *genotype* is the particular set of alleles that the rat possesses at the junk loci in our study: for example it might have alleles A and B at the first locus, G and G at the second locus, and so on. Every island has its own allele frequencies, so the rat with genotype AB/GG will have different belongingness probabilities for every different island – just as a blond man might have a belongingness probability of 0.8 to Sweden and 0.3 to England. If it is known that an island has 60% alleles of type A and 40% of type B at the first locus, and 30% alleles of type G at the second locus, our rat's log genotype probability for this island would be $\log\{(2\times0.6\times0.4) \times (0.3\times0.3)\}$. The contribution for the first locus is multiplied by two because the AB alleles could have arisen two ways, either getting A from the mother and B from the father, or the reverse. These calculations rely on two assumptions: firstly that each locus is in *Hardy-Weinberg Equilibrium (HWE)*, so that the genotype probabilities for the locus can be obtained by multiplying the allele probabilities as above; and secondly that the loci used are independent (described as *linkage equilibrium*), so that the probabilities for different loci can be multiplied together. Some loci may have to be discarded if tests indicate that there are substantial deviations from Hardy-Weinberg or linkage equilibrium.

In practice, we will not know that the island has exactly 60% allele A, 40% allele B, and so on. These numbers have to be estimated from the animals caught on the island. The sampling error in estimating these frequencies is accommodated in the log genotype probabilities, so the approach is not quite as simple as inserting the sample frequencies 0.6 and 0.4. In particular, if an allele C. is not sampled on island 1, it doesn't mean it is absent there. The log genotype probabilities account for the possibility that allele C. might be present at low frequency, and will not completely exclude the island as a possible source for a rat with allele C. This is accomplished through a Bayesian method, so the belongingness probabilities are sometimes called *log posterior genotype probabilities*, with *posterior* indicating that they are the probabilities obtained after the allele frequencies have been estimated. The methods we use for belongingness computation are identical to those found in the free program GeneClass2 (Piry *et al.* 2004), using the Bayesian criterion of Baudouin and Lebrun (2001).

Given a rat's observed alleles, we can calculate the genotype probabilities, or belongingness probabilities, for each of the possible source islands in our study. A powerful way of conveying this information is to plot it on a graph. If there are two possible source islands, we plot the belongingness probabilities for the two different islands on a two-dimensional scatter-plot, where each point gives the two belongingness probabilities for a single rat. We have found this visual method to be an effective way of communicating genetic structure quickly and easily. It requires an imputation method for dealing with missing genetic data, described in Russell *et al.* (2010). We omit from the plot any rats with missing data at more than three loci. If there are more than two populations, a multivariate plotting method is required, which we do not show here.

If the rat's origin is unknown, for example it has been detected on an island following an eradication attempt, we can *assign* it to a possible source population by selecting the population for which it has the highest belongingness probability. This is akin to estimating that all blond men seen in Zanzibar come from Sweden, on the basis that Sweden has the highest proportion of blonds in the world, so the interpretation should be treated with caution. This is why we recommend the visual approach, which might reveal that the blond man has an excellent fit to Sweden but also a perfectly reasonable fit to England. Nonetheless, it is useful at times to collapse the findings to a single selected population source. The population to which the rat has the highest 'belongingness' is given by the percentage scores output by GeneClass2.

We test for deviations from Hardy-Weinberg and linkage equilibrium using Genepop on the Web (Rousset 2008). For Hardy-Weinberg proportions, we use the option for an exact test when there are fewer than five alleles at a locus, and for the remaining loci we use Guo and Thompson's (1992) unbiased estimate of the exact p-value.

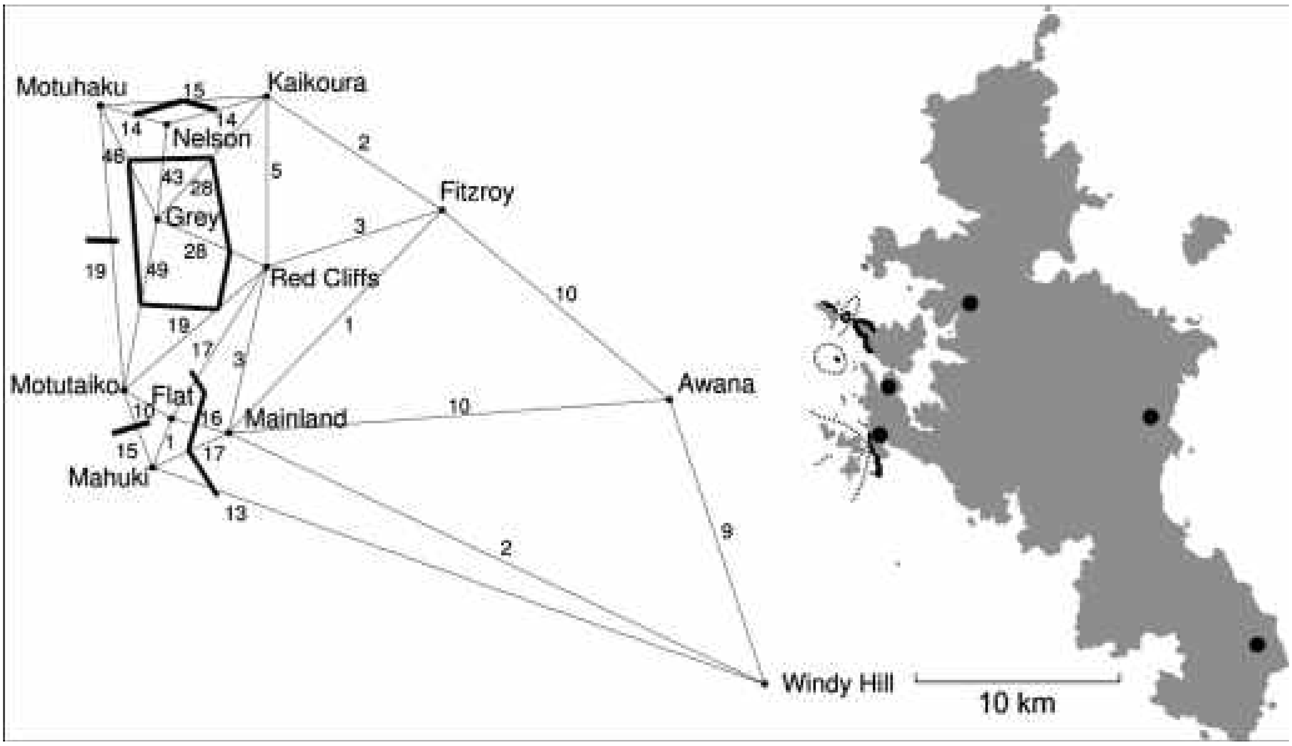

Fig. 2 Genetic relatedness network. Numbers on the lines are F_{ST} values multiplied by 100 and rounded to the nearest integer. Values of 13 and above constitute a genetic 'boundary', marked by thick lines. The map on the right shows the physical locations of the genetic boundaries (four dashed lines). Cliff regions are marked on the map with bold black lines.

Keeping out the neighbours: genetic boundaries, cliffs and water crossings

Using the tools of genetic distance (F_{ST}) and belongingness, we can investigate associated geographical features. We construct a genetic relatedness diagram for F_{ST} and search for genetic boundaries using the Monmonier algorithm (Monmonier 1973) from the package ADEGENET (Jombart 2008) in R (R Development Core Team 2009). The Monmonier algorithm finds the pair of islands with the highest F_{ST} between them, and grows boundaries until it can no longer find island pairs with an F_{ST} above a pre-set threshold, which we set at 0.13.

Using the genetic relatedness diagram, we determined a *separation type* between each pair of islands on the diagram, on the basis of maps, aerial photographs, and fieldworker reports. If islands are separated by a water gap of 1km or more, their separation type is recorded as 'long water'. For gaps of less than 1km, the type is recorded as 'cliff' if the separation is severely cliffy or otherwise inaccessible on one or both sides of the crossing, and 'beach' otherwise. The other separation types are 'land' if the locations are connected by land, even if the distance is considerable; and 'none' when assessing belongingness for a rat into its own population.

We can investigate the impact of separation type on both F_{ST} and belongingness. For F_{ST} we plot the pairwise F_{ST} estimates according to separation type. For belongingness, we conduct a simple linear regression with response of log genotype probability for every rat into every population in the network, and predictors given by two categorical variables, the first being separation type between the rat's sampling population and the target population, and the second with a different level for each target population. The results of interest are the estimated levels for the different separation types: beach, cliff, long water, and land, which show the impact of separation type on belongingness probability.

RESULTS

Genetic boundaries

Genetic boundaries plotted onto the map (Fig. 2) visually appear to correlate with long water crossings and cliffs. In particular, there are strong genetic boundaries

Table 1 Summary statistics for genetic assignment analyses shown in Fig. 4. H_e , H_o and Hardy-Weinberg tests were calculated using Genepop on the Web (Rousset 2008). An individual is heterozygous at a locus if its two alleles are different. H_e gives the mean (across loci) of the proportion of individuals that would be heterozygous at that locus under Hardy-Weinberg equilibrium, and H_o gives the equivalent mean proportion observed in the sample. The HW exact test has null hypothesis that the genotype proportions are in HWE, and alternative hypothesis that they are not.

	Kaikoura	Mainland	Broken Islands
Number of rats included	60	54	60
Mean number of distinct alleles per locus	7.7	8.4	5
Expected heterozygosity, H_e	0.71	0.71	0.57
Observed heterozygosity, H_o	0.65	0.67	0.53
p-value for HW exact test	0.07	0.34	0.48

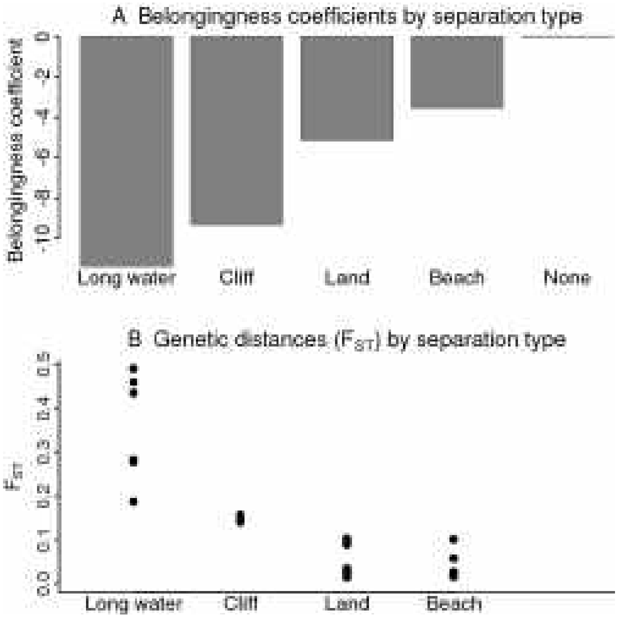

Fig. 3 Impact of separation type on belongingness coefficients (top) and F_{ST} (bottom). For belongingness coefficients, the most negative effects suggest the greatest barriers to genetic relatedness. For the genetic distance F_{ST}, the most positive values give the greatest barriers. Both measures give the same ordering of separation types.

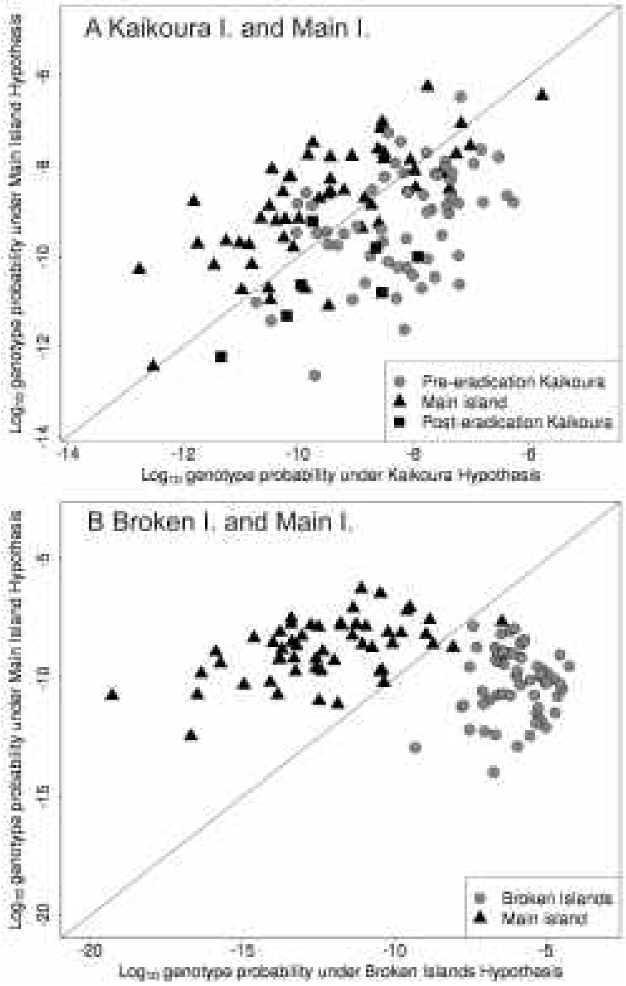

Fig. 4 Belongingness diagram for Main Island rats, sampled from Fitzroy, Red Cliffs, and Mainland areas on Fig. 1, against (A) Kaikoura rats, and (B) Broken Island rats.

between the tiny Grey Group Islands and all other locations, corresponding to long water crossings. There are also clear boundaries along the cliffy areas from the main island (Aotea) to the Broken Islands, from Kaikoura to Nelson, and from Nelson to Motuhaku. The beach crossings between Kaikoura and Fitzroy / Red Cliffs areas, and between Motutaiko and Flat Islands, and Flat and Mahuki Islands, are of similar sizes to the cliffy crossings but do not present genetic boundaries.

We also calculated genetic distances, and belongingness coefficients from the regression, categorized by separation type (Fig. 3). Both methods reflect the same picture: long water crossings create the largest genetic boundaries, followed closely by cliff crossings, then land and beach separations represent substantially less genetic difference.

Survivors or reinvaders?

The exact test for Hardy-Weinberg equilibrium indicated significant departures from equilibrium at three loci: D10Rat20, D20Rat46, and D5Rat83. For a conservative approach, we present results with these three loci excluded from assignment analyses; however, there are no substantive changes in our conclusions when these loci are included (see also Table 1).

The linkage disequilibrium tests revealed only minor evidence of linkage disequilibrium among the Broken Island rats.

Rats were eradicated on Kaikoura, Nelson, Motuhaku, and Grey Group in August 2008. New rats were caught in traps on Kaikoura from March 2009 onwards, and a total of 11 rats and two mice (*Mus musculus*) were caught up to November 2009 and submitted for DNA testing. One of the rats was discovered by DNA analysis to be Pacific rat (*Rattus exulans*). Neither mice nor Pacific rats had been detected on Kaikoura before the eradication, despite 61 ship rats being trapped from 2005 - 2008. If they were present before the eradication, these species might have been undetected due to competition for bait from the more dominant ship rats. Pacific rats and mice are considered unlikely to be swimmers, so their post-eradication presence suggests either survivors of the eradication, or transport by boat. Of the remaining ten ship rats, eight were fresh enough when preserved to provide good DNA.

Of the eight post-eradication Kaikoura rats, one had a strong assignment to the Broken Islands. Its belongingness score for the Broken Islands was in the centre of those from genuine Broken Islands rats. Only four of the 211 rats sampled from outside of the Broken Islands in 2005-2008 equalled or surpassed this score (none if all ten loci were used). This presents very strong evidence that this rat came from the Broken Islands. The distance is too far for swimming, and implies boat transport.

Each of the remaining seven rats were given two belongingness probabilities (log-genotype probabilities) identified in Fig. 4a: one for the hypothesis that it is a survivor from Kaikoura Island, the other for the hypothesis that it came from the main island (Aotea), grouping together the locations Fitzroy, Red Cliffs, and Mainland from Fig. 1. Circles on the plot denote rats sampled on Kaikoura before the eradication from 2005-2008. Triangles denote rats sampled in the aforementioned three mainland sites in 2005-2008. Squares denote the post-eradication rats whose source we wish to determine. A high value on either axis represents a good fit to the corresponding population, and the diagonal line represents an equally good fit to both.

We found a large overlap between the two populations, in keeping with the low F_{ST} values and accessible landings on Kaikoura. This makes it very difficult to distinguish between the survivor and reinvader hypotheses for these rats. However, six of the seven rats fell below the diagonal line (Fig. 4a), favouring the hypothesis that they are survivors of the eradication from Kaikoura. Although the hypothesis of swimmers from Aotea cannot be excluded for any of these rats individually, it is extremely unlikely (p=0.001) that a group of seven swimmers would yield six or more with a better belongingness to Kaikoura than to their native Aotea. Thus we have very strong evidence that these seven rats include some survivors of the eradication.

The leftmost of the post-eradication rats in Fig. 4(a) has a poor fit to all our sampled populations on Great Barrier Island, having the worst all-round fit out of all 270 rats we have sampled in the archipelago. This raises the possibility that it might have arrived by boat from outside the region.

The Broken Islands are separated from Aotea by rugged terrain on the Aotea side. By contrast with Kaikoura, the plot for the Broken Islands (Fig. 4b) clearly distinguishes between rats from the Broken Islands and those from Aotea, even though the water gap is less than 300m. We thus have much greater power to discriminate between survivors and reinvaders for the Broken Islands case, should new rats be detected. The plot, and the statistics in Table 1, indicate that Broken Islands genetics form a subset of Aotea genetics, in the sense that Broken Islands rats largely have a good fit to the Aotea population (i.e. circles have a high score on the vertical axis in Fig. 4(b)), but Aotea rats do not have a good fit to the Broken Islands population, shown by the low scores of triangles on the horizontal axis of Fig. 4(b).

DISCUSSION

Our results from Great Barrier Island suggest that cliffs may be a significant factor in limiting gene flow for ship rats between two islands over short water crossings. Ship rats are capable climbers, so this is perhaps a surprising result. There are many possible behavioural reasons for cliffs to act as boundaries. However, it is also possible that the cliffs on Great Barrier Island are not the cause of the separation, but are simply associated with some other factor, such as water currents.

The genetic results from post-eradication Kaikoura Island, together with an unexpected Pacific rat and two mice, provide strong evidence that in early 2009 there were survivors of the August 2008 eradication. While disappointing, we do not know how unusual this result is, because there is often no post-eradication monitoring until two years after the eradication has taken place. No evidence of breeding was found among the post-eradication rats in early 2009. At least one other rat was almost certainly transported by boat from the Broken Islands. The genetic diagrams show that it will be a challenge to keep Kaikoura rat-free, and that we cannot be conclusive in discriminating between survivors and swimmers. Some threats have been removed by the additional eradications that took place in 2009 on the Broken Islands and the main island. Future risk can be reduced by publicity among boat users in the area, and further control on the mainland fringe.

DNA profiling can be a powerful tool in conservation management, both for understanding underlying behaviour and for sourcing individual rat invaders. To best exploit the opportunities offered, coordination is needed among different management and research groups. Genetic results from different labs are only comparable if they use the same genetic loci and share control samples for calibration. The ideal would be to collate genetic results from around the country into a national database, accessible to any management groups with reinvaders to source. Crucially, we encourage managers to take DNA samples before any eradication is attempted. Studies should aim for samples of at least 30 rats from each source population, including the island for eradication, although some islands will provide a strong genetic signature with fewer samples.

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Advantages and challenges of government, non-profit and for-profit approaches to eradications: leveraging synergies by working together

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Abstract The removal of invasive mammals from islands has become a powerful tool for restoring ecosystems and preventing extinctions. As larger and more complex islands are being targeted for restoration, eradication campaigns will become even more complex and multidimensional – biologically, operationally, and financially. Eradication projects are typically conducted by governmental conservation agencies (GCAs), non-governmental organisations (NGOs), or for-profit enterprises (FPEs). Partnerships across these three organisational types are increasingly common. The organisational structure of the institutions involved in eradication and other restoration campaigns undoubtedly plays a role in the effectiveness and nature of outcomes. We briefly explore the advantages and challenges of different organisational structures conducting invasive mammal eradication programmes. We do so to explore potential synergies that arise from strategic partnerships between different types of organisations. GCAs commonly enjoy special privileges, reliable operational budgets, and simplified lines of communications – all of which are advantages to managing an eradication project. However, they often face challenges, including lack of experience, vulnerability to outside pressures, and a risk averse atmosphere. NGOs often have relative advantages in fundraising capacity and flexibility. Their challenges include permitting, fundraising pressure, and less accountability. FPEs commonly enjoy less regulation and bureaucracy, have more operational flexibility and excellence, and incentives for innovation. Limited project control, near-sighted investment, and risk avoidance can present them with challenges during eradication projects. Recent partnerships that executed watershed eradication campaigns over the last decade suggest that working together on island restoration programmes can leverage synergies. Partnering across organisational structures is likely to be a highly effective strategy for mainstreaming invasive species eradications.

Keywords: Innovation, partnerships, organisational structure, mainstreaming eradications, Project Isabela, NZ Department of Conservation, Santa Cruz Island, Anacapa Island

INTRODUCTION

The removal of invasive mammals from islands has become a powerful tool for restoring ecosystems and preventing extinctions. There have been over 900 successful eradications worldwide, and recent innovative programmes suggest that area is often no longer the limiting factor for removing invasive mammals from islands (Cruz *et al*. 2009; Donlan and Wilcox 2008; Howald *et al*. 2010; Macdonald and Walker 2008; McClelland and Tyree 2002). Eradication projects are complex endeavours that blend logistical planning, environmental compliance, scientific research, operational management, and public relations. As larger and more complex islands are targeted for restoration, eradication campaigns will become even more complex and multi-dimensional – biologically, operationally, and financially.

Eradication projects are typically conducted by governmental conservation agencies (GCAs), non-governmental organisations (NGOs or, in a few cases, community groups), or for-profit enterprises (FPEs). Partnerships across these three organisational types are increasingly common. The organisational structure of the institutions involved in eradication and other restoration campaigns undoubtedly plays a role in the effectiveness and nature of outcomes. This effect of organisational structure on outcomes is seen in other disciplines. For example, in primary health-care, large managed care organisations often fail to provide quality care due to complexities and fragmentation of the organisation (Barr 1995). In contrast, smaller organisations often lack the internal depth and external reach to drive objectives through complex bureaucracies.

In this paper, we briefly explore the advantages and challenges of different organisational structures conducting invasive mammal eradication programmes (i.e. GCAs, NGOs, and FPEs). Our objective is to explore potential synergies that arise from strategic partnerships between different types of organisations. We highlight some of those advantages, challenges, and synergies by briefly discussing four recent eradication programmes as case studies.

THE ROLE OF ORGANISATIONAL STRUCTURE

An organisation emerges whenever people cooperate over time in order to get things done. An effective organisation is one that is able to achieve its purposes or aims. Four factors influence an organisation's effectiveness: system, culture, leadership, and power (Fairtlough 2005). Organisations garner advantages when systematic and standard procedures are in place. Organisations with many established systems and standards are bureaucratic, and are often viewed as cumbersome. Systems and rules, however, can promote effectiveness. A shared organisational culture encourages efficient communication within an organisation. Similar to its systems, an organisation's culture can be either enabling or coercive (Fairtlough 2005). A leader makes sense of an organisation and helps others do the same. Research in the private sector has revealed insights on superior leadership and its characteristics, perhaps most importantly the requisite of a combination of personal humility with professional will (Collins 2005). Power, both complex and dynamic, is a necessary part of getting things done. More often than not, discourse about power within organisations is suppressed, and a hierarchical nature of power is considered to be inevitable and natural (Hardy and Stewart 1996). Heterarchial organisations are more horizontal in nature and can hold advantages over those with more hierarchical structure, such as speed of action

Pages 432-436 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

(Fairtlough 2005). Effective organisations tend to possess enabling systems, trust-generating cultures, superior leadership, and accountable power (Collins 2001, 2005; Fairtlough 2005).

The eradication of invasive mammals from islands has become highly specialised and often relies heavily on technology and skilled labour. In the private sector, organisations that specialise in a few complex operations often have an efficiency advantage over less specialised organisations (Collins 2001). Such efficiency advantages are also likely to apply to conservation organisations, including those that specialise in the eradication of invasive mammals (Roemer and Donlan 2005).

Organisational Structure and Island Restoration

Government Conservation Agencies: When and where GCAs are committed to eradications, there have been highly successful programmes, such as the 50+ year commitment to fox eradication from the Aleutian Islands, Alaska by the U.S. Fish and Wildlife Service (Ebbert and Byrd 2002). In fact, most invasive mammal eradication campaigns have been conducted by GCAs, particularly those in Australia and New Zealand (Campbell and Donlan 2005; Howald *et al*. 2007). The natural heritage of both countries has long been heavily impacted by invasive species, and thus agencies exist that have invasive species research and management as one of their primary roles. Examples include the New Zealand Department of Conservation (DOC) and Australia's Invasive Animals Cooperative Research Centre.

There are some clear advantages of GCAs conducting eradication campaigns. Many GCAs enjoy special privileges that facilitate the efficiency of an eradication campaign, such as the ability to use select toxins or exemptions from permits. They may also have reliable operational and programmatic budgets that can be used to subsidise costly components of eradication campaigns such as logistics, legal council, and environmental monitoring. Further, a GCA eradication campaign may enjoy simplified lines of communication and require less inter-agency communication. This is particularly true in countries like New Zealand, where there is a single layer of bureaucracy compared to countries with multiple layers of government (e.g., provincial and federal).

Other common characteristics of GCAs present challenges to operating effective eradication campaigns. Given the multi-layer, largely hierarchical decision-making infrastructure of many government agencies and their adherence to internal policy, they can face challenges when decisions need to be made swiftly. Many GCAs around the world have little, if any, experience with invasive species management, which presents a suite of challenges for managing an eradication campaign. GCAs also tend to be risk averse and subject to political and public opinion pressures – inside and outside a particular project (Roemer and Donlan 2005).

Non-governmental Organisations: NGOs are increasingly playing prominent roles in island restoration programmes. For example, the NGOs Island Conservation (USA) and Grupo de Ecología y Conservación de Islas (México) have made impressive strides in restoring the islands of northwest México over the past fifteen years (Tershy *et al*. 2002; Aguirre-Muñoz *et al*. 2008). An NGO conducting an eradication programme may enjoy some advantages. First, NGOs, particularly non-membership organisations, possess systemic flexibility with respect to prioritisation, planning, and operations. Second, they have potential access to more revenue streams via diverse fundraising activities compared to GCAs. Third, the independent nature of NGOs can shelter them from some political and social pressures, allowing them to become embedded in the communities where they are working.

NGOs are by no means immune to the many challenges of eradication campaigns. They can be stifled, sometimes for long periods, by permitting requirements and environmental compliance. This is particularly the case with nebulous or overly onerous permit processes present in some countries. The economics of eradication campaigns and programmes are often complex (Donlan and Wilcox 2007), with short bursts of high activity followed by long periods of little or no activity. This cycle can put financial pressure on NGOs to maintain fundraising abilities in order to maintain capacity for the next eradication campaign. Lastly, the efficiency and effectiveness of conservation NGOs can suffer due to a lack of accountability from funders (Ferraro and Pattanayak 2006). This climate is, however, beginning to change as environmentally focused foundations and others adopt return on investment approaches.

For-profit Enterprises: FPEs are playing increasingly important roles in invasive mammal eradications around the globe, either managing entire eradication campaigns or solely the on-the-ground components (Bell 2002; Kessler 2002; Macdonald and Walker 2008). FPEs are often subject to less regulation and bureaucracy than other organisations. FPEs often also have operational flexibility; they are able to hire the most highly qualified personnel and adopt best practices for the situation under the constraints of the contract. Both of these conditions – less "red tape" and operational flexibility – contribute to promoting innovation. Lastly, appropriately structured contracts, such as those that are performance-based, can promote innovation in techniques and technology.

Being vulnerable to contracts for solvency, FPEs face many challenges to conducting invasive mammal eradication programmes. They are often particularly vulnerable to funding gaps. FPEs can struggle to maintain a highly skilled staff and costly equipment (e.g., helicopters) during the downtime between campaigns. Many challenges stem from the nature of the contract. FPEs can have limited control of a project due to contract restrictions. This effect is compounded by "contract paradox": the largest liability with contracting out eradications is a lack of understanding about the effort and skills needed to achieve eradication by the contractee, which is compounded by the fact there is a lack of suitably qualified contractors with the skills or knowledge to complete the eradication. FPE's may be resistant to performance-based contracts (where time to eradication is the performance measure), since the complex nature of some eradications and the range of issues outside of the FPE's control present a suite of risks. Subsequently, FPE's often prefer "input" contracts, where they are paid for delivered tasks (e.g., helicopter hours or number of treated hectares) that have a chance of providing eradication success. Collaborations that work toward performance-based contracts that also share operational and political risks will increase the conservation return on investment of invasive mammal eradications from islands.

Leveraging Synergies

Eradications are becoming more complex in all respects. Eradication campaigns are increasingly run by partnerships as opposed to a single organisation. We briefly explore four recent landmark eradication campaigns, starting first with the collective accomplishments of DOC.

DOC and Campbell Island Restoration Project (New Zealand): GCA Campaign

In 2001, DOC eradicated Norway rats (*Rattus norvegicus*) from Campbell Island (11,216 ha), the world's largest invasive rodent removal to date (McClelland and

TABLE 1 Innovations within the NZ Department of Conservation (modified from Wright and de Joux 2003).

Mindshift: moving from control mentality to eradication ethic.
Capability Development Across Disciplines: engineering improvements in methods and technologies.
Mainstreaming Best Practices: using the Island Eradication Advisory Group to ensure best practices are employed.
Stretch Goals: taking on bigger challenges.

Tyree 2002; Towns and Broome 2003). This landmark project was successful due to decades of experience by DOC in invasive species eradication and project management. The government agency, along with its accomplishments, serves as a premier case study of innovation in the public service (Wright and de Joux 2003). Throughout its existence, DOC has facilitated and institutionalised innovations that contribute to effective invasive mammal eradication projects (Table 1). Those innovations are of high utility to any organisation tackling island restoration programmes.

Several organisational characteristics contribute to DOC's effectiveness at systematically removing invasive mammals from islands. First, the agency has a clear and focused mission, which is to protect and enhance the environment using two key steps: "expand biodiversity effort" and "minimise bio-security risks". DOC is an integrated conservation service, and in general stakeholder buy-in is limited, as are other competing interests such as outdoor recreation at many sites targeted for conservation. This is in contrast to other government agencies outside New Zealand, such as the US National Park Service (US-NPS) and Australia's State Park and Wildlife Agencies. Second, DOC runs its eradications as a programme as opposed to single, independent projects. For example, Campbell Island was part of a larger five-island restoration programme. As such, this provided some funding flexibility within the programme. Staff positions also had secure funding during the long planning process when levels of effort fluctuated greatly, and were dedicated full-time when actual operations were underway.

One potential challenge of a GCA conducting eradication campaigns single-handedly is the perception of them serving as the prosecutor, defender, judge, and jury. For the Campbell Island programme, this potential transparency issue was overcome by contracting out the legal tests required for resource consent to another territorial authority. In nations were there are well-established federal and state natural resource agencies that provide some "checks and balances", transparency is less of a project risk.

Anacapa Island Restoration Project (USA): A GCA-NGO Partnership

The Anacapa Island Restoration project removed ship rats (*Rattus rattus*) from a small island (300 ha) in the US Channel Islands National Park, located off southern California. The project was the first aerial rodenticide application in North America and involved innovative non-target mitigation strategies due to the presence of an endemic rodent. The eradication campaign and the non-target mitigation programmes were both successful (Howald *et al*. 2010). The Anacapa Island Restoration Project was conducted as a partnership between the US-NPS and the NGO Island Conservation.

This GCA-NGO partnership provided some advantages. The US-NPS managed the political and permitting issues, while Island Conservation focused on the scientific, technical and logistic obstacles. When legal claims were made by animal rights organisations, the government was able to provide the necessary legal resources to successfully fight those claims in court (Howald *et al*. 2010). Island Conservation was not implicated in the lawsuit; US-NPS was both the landowner and held the ultimate decision authority for the project. The division of labour proved effective, allowing the NGO to focus on operations and the GCA to provide critical support on and off the island, essentially shielding the project from the legal and negative public relations campaigns. Effective partnership and dialogue between the partners provided a clear division of responsibilities that led to the successful implementation of the project. The main potential disadvantage of a GCA-NGO partnership, or almost any partnership, is the diffusion of responsibility. This risk must be managed through a clear and effective partnership relationship with absolute transparency about project activities on both sides. Responsibilities need to be clearly defined to ensure success, and so if the eradication fails, it cannot be blamed on one group or another.

Project Isabela (Ecuador): A GCA-NGO-FPE Partnership

Project Isabela was a multi-stage eradication programme in the Galápagos Islands that targeted feral goats (*Capra hircus*), pigs (*Sus scrofa*), and donkeys (*Equus asinus*) on three islands in the archipelago. Two of the islands targeted, Santiago (58,465 ha) and Isabela (458,812 ha), were much larger in size than any other islands where invasive herbivores had previously been removed (Campbell *et al*. 2004; Carrion *et al*. 2007; Cruz *et al*. 2009; Cruz *et al*. 2005). This successful project was funded in part by the Global Environment Facility and co-managed by the Galápagos National Park and the Charles Darwin Foundation. The United Nations Development Program (UNDP) was also a partner, managing funds coming from the Global Environment Facility. In addition, aerial hunting aspects of the campaign were contracted to a private company from New Zealand. The GCA-NGO-FPE organisational structure of Project Isabela provided a number of advantages in navigating the many planning and implementation challenges of the project, particularly the dynamic socio-politics. Project Isabela survived 10 Galápagos National Park Directors, several Ministers of the Environment, and five Ecuadorian Presidents.

The project was embedded between two institutions, which provided it some autonomy and two potential structures for decision-making. This structure allowed for flexibility, drawing from opportunities and benefits from each of the institutions. For example, the Charles Darwin Foundation acted as a conduit for funding, and enjoyed greater flexibility in budgetary spending than did the Galápagos National Park, allowing the Foundation to cover project costs when needed. The eradication campaign was thus able to continue without breaks despite a lack of funding within the National Park for hunters' salaries at the start of each financial year. The non-profit and diplomatic status of the Charles Darwin Foundation also facilitated contracting outside of Ecuador and importation of firearms and ammunition. Project Isabela's formal affiliation with the Galápagos National Park offered distinct advantages as well, including direct access to the Park's infrastructure, which facilitated efficient logistics such as access to boats and dog kennels.

There were also project advantages achieved by contracting out the aerial component of the eradication campaign. Most importantly, highly skilled personnel and specialised equipment could be efficiently folded into the programme, which allowed additional focus and resources to be spent on more experimental components of the project. An aerial hunting contract also provided some risk sharing. While the risk of eradication failure was carried by Project Isabela, since the FPE was paid per unit of effort (i.e. helicopter hours and a mobilisation component), operational delays and any associated financial risks were borne on the FPE as opposed to Project Isabela.

Not surprisingly, challenges also emerged from this GCA-NGO-FPE partnership. There was added complexity in project planning, particularly the need for coordinating annual budgets. Project Isabela was also vulnerable to potential crisis situations within the Galápagos National Park or the Charles Darwin Foundation. During the project, a large-scale oil spill and a large wildfire overwhelmed staff capacity, and temporarily paralysed Project Isabela.

The use of the intergovernmental agency UNDP as a partner in Project Isabela has yet to be replicated in any other eradication campaign. The UNDP provided a project advantage because of the ability to commit to large cash outlays from a single source. The agency also streamlined large contracts (e.g., helicopter contracts), and provided high-level direction and pressure to keep the project a priority for the various Ecuadorian government agencies. The UNDP did, however, bring added bureaucracy that was more focused on process than on products. As a consequence, commitment to timelines was difficult, and some funds were needlessly consumed while the project awaited the release of additional funding.

Santa Cruz Island (USA): A GCA-NGO-FPE Partnership

Feral pigs were recently removed from Santa Cruz Island, Channel Islands National Park. Santa Cruz Island (24,900 ha) is located off southern California and co-managed by the NGO The Nature Conservancy (TNC) and US-NPS. The eradication campaign, its ground operations run by Prohunt, Inc., removed pigs from the island in fifteen months, followed by 11 months of monitoring (Macdonald and Walker 2008; Morrison *et al*. 2007).

The pig eradication on Santa Cruz Island was a result of GCA-NGO-FPE partnership between the US-NPS, TNC, and Prohunt, Inc. Synergy emerged from this partnership that contributed to the unprecedented speed and effectiveness of the eradication campaign. The US-NPS and TNC shared resources to garner support and approval of the eradication programme, including fundraising, environmental compliance, and public outreach. Local, state, and federal political support was critical for the project, and resources and expertise were needed to overcome animal welfare challenges throughout the project. Such support ensured operations continued uninterrupted despite multiple legal challenges. Support and leadership on these issues by the US-NPS and TNC allowed Prohunt to focus on the actual removal of feral pigs from the island.

The structure of the eradication contract with Prohunt, Inc. facilitated an effective campaign. With the exception of helicopter fuel and on-island accommodation, which were supplied by US-NPS and TNC, the three-year fixed price end-user contract was straightforward to manage. As long as the contractor did not violate established guidelines (e.g., poison, snares, and lead ammunition were banned) and followed reporting standards, Prohunt was free to adopt a suite of eradication techniques, along with the sequence in which they were applied.

CONCLUSIONS AND RECOMMENDATIONS

Advantages and challenges to successful eradication projects can depend on the type of organisational structure (Table 2). There can be real challenges to collaborations across organisational types in invasive mammal eradications. For example, different cultures can make effective communication difficult. Further, unless roles and responsibility are explicitly defined, diffusing and sharing responsibilities can present a moral hazard and elevate the risk of eradication failure. Based on our experiences, however, the potential advantages of collaboration are often greater than the challenges. Every eradication campaign is unique, with partners often hoping to move a project forward in their own way. No matter which way a project or programme is moved forward, it is important to leverage those advantages, while minimising the challenges in order to mainstream invasive species eradication from islands. Partnering across organisational structures is an effective strategy for leveraging synergies, and successful implementation of island restoration projects.

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TABLE 2 Some advantages and challenges of eradication campaigns run by government agencies, non-governmental agencies, and for-profit enterprises.

	Govt Conservation Agency	**Non-governmental Agency**	**For-profit Enterprise**
Advantages	Special privileges	Fundraising capacity	Less regulation and bureaucracy
	Reliable operational budget	Flexibility (planning & operations)	Incentive for innovation
	Hierarchial and/or simple lines of communication	Independence	Operational flexbility and excellence
Challenges	Lack of experience	Permitting	Risk avoidance
	Subject to political pressures	Fundraising pressure	Limited project control
	Risk averse	Less accountability	Near-sighted investment

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Behaviour of invader ship rats experimentally released behind a pest-proof fence, Maungatautari, New Zealand

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Abstract Six ship (roof, black) rats (*Rattus rattus*) were cage-trapped adjacent to a pest-proof fence and released with radio transmitters inside the 65 ha pest-free exclosure at Maungatautari, North Island, New Zealand, to mimic reinvasion. Unexpectedly, four of the six rats climbed back out of the exclosure and returned to their original home ranges after periods ranging from a few hours to seven days. All six rats travelled along the fence top at some time during follows, and only three of the six used tracking tunnels set on a 50 m grid inside the exclosure to detect invaders. The rats that remained inside the fence stayed within C. 100m of the release point for about three days, then made increasingly large (to 1100m) movements into the reserve. Resultant range lengths greatly exceeded those of four other rats radio-tracked outside the fence where rat density was higher. This behaviour is very similar to that reported for experimentally released house mice (*Mus musculus*) and Norway rats (*R. norvegicus*) on islands. These results suggest that a) some invading ship rats may themselves vacate a fenced sanctuary without encountering efforts to detect and remove them; b) rats at low density have much larger movements than occur in home ranges at typically higher mainland densities, and c) managers should target rat invaders with detection and killing devices within 100 m of a fence breach for at least three days, and some traps should be set on top of the fence.

Keywords: *Rattus rattus*, rodent invasion, rodent behaviour

INTRODUCTION

Mammal pest eradications have been achieved on 432 islands around the world, 133 of which are in New Zealand (Clout and Russell 2006; Global Island Invasive Vertebrate Eradication Database, accessed 29 January 2010). The same eradication opportunities are increasingly being created in mainland situations by the use of pest-exclusion fences (Day and MacGibbon 2007) which provide a barrier to pest reinvasion, as water does in island situations. The largest fence-protected area is at Maungatautari in the central North Island, New Zealand, where 3400 ha of forest now forms a mainland island protected by a 47 km fence that can exclude all introduced mammals including mice (*Mus musculus*).

However, reinvasion is always possible in both marine (Russell and Clout 2005) and mainland islands. In the latter case, pests on the outside may jump in from overhanging branches; be thrown in when trees on the outside crash over the fence; climb over branches or tree-fern fronds that temporarily fall against the fence; walk in through holes caused by falling trees, errant vehicles, water scouring or hunters' bullets, or even be carried in by birds of prey.

In New Zealand, ship rats (*Rattus rattus*) are very widespread, abundant (typically 2–6 per ha in podocarp-broadleaved forest), frequently arboreal (Innes 2005), and likely reinvaders of fenced sanctuaries. In a previous Maungatautari study, nine pest mammal species were filmed exploring fake breach holes in the fence at ground level; ship rats were the second-most frequent visitor after mice, entering holes every two nights in summer and every four nights in winter (Connolly *et al*. 2009). Also, on average 9.4 rats per night were filmed travelling along the gutter on the underside of the fence hood 2m above ground, suggesting that several rats per night may find any holes in the hood (Connolly *et al*. 2009).

The behaviour of invader rats is little understood. Russell *et al*. (2008, 2010) released and radio-tracked male Norway rats (*R. norvegicus*) on islands but we know of no similar studies with ship rats or in fenced sanctuaries in mainland situations. Learning more about invader behaviour can help guide managers to more effective detection and removal strategies and techniques.

We aimed to determine reinvasion behaviour of ship rats by mimicking invasion events from existing home ranges. We cage-trapped adult male ship rats adjacent to a pest-free reserve at Maungatautari, attached radio transmitters to them, and released them again on the inside of the pest-proof fence to observe behaviour. We also radio-tracked three females and a male ship rat outside the fence to trial the transmitters and rat-tracking procedures, and to look at movement behaviour in the 'source' population at this site.

Russell *et al.* (2010) noted that movements of Norway rats released on marine islands were generally random but showed a tendency to return to previously used den sites, and suggested that continually increasing range size would effectively enable colonising individuals to rapidly find mates in a new environment. We predicted these same outcomes for ship rats; in particular, that individuals would move very large distances inside the reserve to look for other conspecifics.

METHODS

Site description and management

Maungatautari is an eroded andesitic volcanic cone in the central North Island, New Zealand, southwest of Hamilton City. The 3400 ha forest in the reserve is dense, primary, podocarp-broadleaved forest dominated on its lower margins by scattered large rimu (*Dacrydium cupressinum*) and northern rata (*Metrosideros robusta*) over a canopy of tawa (*Beilschmiedia tawa*), mangeao (*Litsea calicaris*), hinau (*Elaeocarpus dentatus*), miro (*Prumnopitys ferruginea*), rewarewa (*Knightia excelsa*) and pukatea (*Laurelia novae-zelandiae*) (Burns and Smale 2002).

The Maungatautari Ecological Island Trust (MEIT) was formed in 2002 to ring-fence the mountain and to eradicate all mammal pests inside the fence. Its vision is "to remove forever, introduced mammalian pests and predators from Maungatautari, and restore the forest to a healthy diversity of indigenous plants and animals not seen in our lifetime" (McQueen 2004). In 2004, two smaller exclosures were

Pages 437-440 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

fenced and cleared of mammals as pilot programmes prior to fencing and eradicating invasive mammals on the entire mountain two years later. We worked in one of these, the 65 ha 'southern exclosure', that is surrounded by an Xcluder pest-proof fence (Day and MacGibbon 2007) and bordered by native forest and farmed pasture. The last resident ship rat was trapped in the southern exclosure in March 2006 (Speedy *et al*. 2007), a month before we started working there.

The Xcluder fence used at the Southern Enclosure is based on 2m high posts covered with fine mesh (6 mm x 25 mm). The mesh includes a 300 mm wide horizontal skirt buried 50 mm under the ground and a steel hood around the top of the mesh sloped to the outside of the fence and rolled into a gutter along the outer edge. Animals on the inside of the fence can climb to the top and jump out, but animals trying to climb up the fence from the outside are prevented from reaching the top by the hood and gutter (Day and MacGibbon 2007).

Individually labelled Black Trakka footprint tracking tunnels occur on a 50 x 50 m grid throughout the entire Southern Enclosure, and by maintaining knowledge of the closest tunnel, enabled adequately detailed mapping of the rat locations through the night. Tracking tunnels were baited monthly and are only really effective when baited (Gillies and Williams Unpubl. report), although cards remained in place throughout the research.

Ship rat capture, transmitter attachment, re-release and radio-tracking

Ten ship rats were cage-trapped, anaesthetised and transmitters attached adjacent to but on the outside of the predator-proof fence during February-September 2006 (Table 1). Four rats were released at the point of capture (still outside the fence) to trial radio-tracking techniques and to examine range size in the unmanaged ship rat population; six rats were subsequently released inside the fence immediately adjacent to their capture point, thus mimicking a natural invasion.

Cage traps were baited with peanut butter and carrot, set in the late afternoon and checked that night starting about an hour after dark. Large, mature (testes scrotal), male rats were selected for release inside the fence to avoid the risk of introducing pregnant females into the pest-free reserve, and to ensure that the individuals had sufficient bodyweight to carry a transmitter with ethical safety (Kenward 2001). Mean rat weight was 167 g (range 147–196), so that the 4.5 g transmitters were 2.3–3.0% of rat body-weight. Each rat was released from its cage into a large plastic bag, then anaesthetised with an isoflurane-oxygen mix dispensed via a veterinary Stephen's vaporiser. Transmitters supplied by Sirtrack Ltd used CEPX76 batteries and were fitted as neck collars with a brass loop that functioned as an aerial. Mean anaesthesia time was 11 minutes.

All rats were released as soon as they recovered fully, four at the point of capture, and six inside the fence immediately adjacent to the capture site. Rats were not followed further on their capture night to allow them to find food and shelter.

Subsequent radio tracking was with Telonics TR4 receivers and Yagi aerials, usually with two observers, from late afternoon (to get initial den site) to C. 0100 am the following morning, for 1-16 nights which were not necessarily consecutive (Table 1). We followed rats at C. 40m range and estimated their locations by triangulation. Rat locations were recorded at half-hour intervals against the labelled 50m grid of tracking tunnels.

RESULTS

Movement behaviour

The rats that remained inside the fence stayed within C. 100 m of the release point for about three days, then made increasingly large (to 1100m) movements into the reserve, as predicted (Table 1). They tended to use the same den sites and travel routes for 2–3 nights at a time, then changed both for another 2–3 nights at a new location. Of the two rats that stayed longest inside the exclosure, one died after eating brodifacoum poison bait laid for mice, and the second probably died of a lung infection (perhaps exacerbated by its anaesthesia) although it had also eaten the poison bait laid for mice.

Unexpectedly, four of the six rats released inside the pest-proof fence climbed back out of the exclosure and returned to their original home ranges after periods ranging from a few hours to seven days (Table 1).

Table 1 Gender, release locations, and movement behaviour of ten ship rats radio-tracked at Maungatautari, central North Island, New Zealand.

Gender	Date trapped (2006)	Released inside fence?	Nights of radio-tracking (date span)	Range length (m)	Time in exclosure	Tracking tunnels tracked	Rat fate
M	8 Feb.	no	5 (9-13 Feb)	320	-	-	Remained outside fence
F	9 Feb.	no	4 (10-13 Feb)	90	-	-	Remained outside fence
F	6 March	no	3 (9-11 March)	50	-	-	Remained outside fence
F	6 March	no	3 (9-11 March)	50	-	-	Remained outside fence
M	20 April	yes	7 (21-27 April)	600	7 days	2	Died inside fence from lung infection
M	10 July	yes	16 (11-31 Jul.)	1100	31 days	7	Died inside fence from brodifacoum laid for mice
M	18 Aug.	yes	2 (19-21 Aug)	20	3 days	0	Trapped outside fence in original home range
M	6 Sep.	yes	6 (7-18 Sep)	600	7 days	8	Returned to original home range outside fence
M	21 Sep.	yes	1 (22 Sep)	20	< 24 hrs	0	Returned to original home range outside fence
M	26 Sep.	yes	1 (27 Sep)	20	<6 hrs	0	Returned to original home range outside fence

Range lengths of the three female rats released outside the fence averaged 63 m while the outside male's range length was 320 m (Table 1). Home ranges revealed by radio-tracking included the original cage capture site.

Rat activity

The radio-tracked rats were active only at night, emerging from dens at dusk. We seldom saw the rats while tracking them. Occasionally we observed them up trees and on vines but on too few occasions to reliably report the proportion of time spent above the ground. All six rats travelled along the fence top at some time while being followed.

Den sites were sometimes in logs on the ground, and sometimes up to 15 m above the ground, including in treefern (*Cyathea* and *Dicksonia* spp.) crowns, supplejack (*Ripogonum scandens*) tangles and epiphyte (*Collospermum hastatum* and *Astelia solandri*) clumps in a variety of canopy and emergent trees. Some rats used the same dens for several nights in a row and some changed dens every night. Newly released rats tended to use the same den for 2–3 nights when they were first placed inside the fence.

Use of tracking tunnels

All three of the six released rats that remained inside the exclosure for more than three days used the tracking tunnels set to detect survivors and invaders (Table 1).

DISCUSSION

Movement behaviour

The rats that remained inside the fence stayed within C. 100 m of the release point for about three days. This is much greater than the time that may be expected as a behavioural response to handling and anaesthesia (Russell 1983). They then made increasingly large (to 1100 m) movements into the reserve. This behaviour is very similar to that reported for experimentally released house mice (MacKay 2011) and Norway rats (Russell *et al*. 2008, 2010) on islands. Russell *et al.* (2010) noted that Norway rats released on marine islands tended to return to previously used den sites, and continually increased their range size, perhaps to enable colonising individuals to rapidly find mates in a new environment. Our experimentally released ship rats showed similar behaviours, often taking similar travel routes from den sites on different nights. Male ship rats in a low density population tracked in beech forest in the South Island, New Zealand, also moved large distances (up to 700 m; Pryde *et al*. 2005), suggesting that an inverse relationship between density and movement is the norm for this species.

Rats released outside the fence at their capture sites showed expected range sizes for their genders (Innes 2005). They were tracked at a different time of year and for fewer days than rats inside, but our conclusion that rats inside the fence moved unusually large distances does not rest primarily with this comparison. Mean range lengths (maximum straight-line measurement within a home range) in Puketi Forest, Northland, were 185 m for females and 159 m for males (Dowding and Murphy 1994); at Rotoehu, central North Island, mean female range length was 103 m and mean male range length was 194 m, and in the Orongorongo Valley, Wellington, maximum range length was 100 m for females and 150 m for males (Daniel 1972). The actual density of rats outside the fence at the time of our study was unknown, but typically is 2–6 rats/ha in North Island podocarp-broadleaved forest (Innes 2005), and we cage-trapped many other rats at the time of initial capture, consistent with this.

The result that most rats returned to where they came from was unexpected, considering the absence of predators and abundant food inside the fenced reserve. Perhaps the rats innately begged sociality of some kind, especially access to mates, or perhaps their knowledge of safe den sites and good feeding places in their original home range was preferable to finding new solutions to these requirements inside the fence. The rats that we released after mid-September left the exclosure within a day, which may be related to the spring onset of the breeding season (Innes 2005). Pest-proof fences are designed to keep animals out, not in, and so are readily scaled from the inside. It is likely that fences 'export' many mobile pests out of sanctuaries in the early stages of their construction and before eradication poisoning commences.

These preliminary results suggest that single invading ship rats may be less of a threat to fenced sanctuaries than previously thought, because most rats apparently do not want to be there. However, our sample size was small, with releases of only one gender and through only part of a year, and it would be dangerous to assume that all ship rats or all species will behave this way. It is conceivable that a rat will invade temporarily – leaving footprints at a tunnel inside the fence and then climbing out. However, it would be impossible to verify this, or to tell the difference between a rat climbing out compared to going further in to the reserve, unless some device records the rat again inside the reserve. In essence this is a repeat of the conundrum about how to verify that an eradication has been successful (Solow *et al*. 2008): how do you confirm the absence of something?

The research needs to be repeated with subadult rats that may be naturally dispersing, with no established home range to return to, and perhaps with female ship rats. What if a male and female arrived together? What if a rat was taken from a faraway location and put in (as could happen with deliberate malicious reintroduction)? And how would cats (*Felis catus*), mice, stoats (*Mustela erminea*), brushtail possums (*Trichosurus vulpecula*) and the other introduced mammal pests in New Zealand behave after they entered a fence breach?

Rat activity and use of tracking tunnels

Basic ship rat behaviours that we observed such as nocturnality, arboreality, and denning were all consistent with previous knowledge (summarised in Innes 2005), although rats on Taukihepa Island with abundant burrowing seabirds frequently denned underground (Rutherford *et al*. 2009). Tracking tunnels successfully detected all three rats that remained inside the exclosure for more than three days but tracking did not closely indicate the extent of rat movement, even with tunnels at 50 m spacing. The tracking technique is clearly useful provided tunnels are kept baited, but an invader ship rat in a very low density population may be hundreds of metres away from the tracked tunnel by the time the card is located. Daily clearance of tracking cards and immediate placement of traps and poison stations will increase the chances of intercepting invaders with killing devices.

RECOMMENDATIONS FOR MANAGERS

This research suggests that where there is a known fence breach through which ship rats may have invaded:

1. Traps should be set on top of the fence as well as on the ground inside, because all rats travelled some distance along the fence top, sometimes hundreds of metres on consecutive nights.

2. Tracking tunnels, traps and poison stations should target rats within 100 m of the breach site for three days after the breach. After this, these devices should be maintained, but the detection net should be substantially broadened in case the rat has moved elsewhere. Two of our experimentally released rats were 600–800 m away from their release point after seven days, and one was 1100 m away after 11 days.

3. Tracking tunnels at 50 m spacing seem to be effective at detecting invader rats provided that the tunnels are baited and checked regularly and that the rats are resident for more than a day or two. Daily clearance of tracking cards and immediate placement of traps and poison stations will increase the chances of intercepting invaders with killing devices before the rat moves on.

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Tawharanui Open Sanctuary – detection and removal of pest incursions

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Abstract Tawharanui Regional Park north of Auckland, New Zealand, is being developed as an Open Sanctuary integrating conservation, recreation, and farming operations. Through a council and community partnership, 550 ha of the Tawharanui peninsula was isolated with a 2.7 km coast to coast pest proof fence, completed in 2004. Multi-species mammalian pest eradication was undertaken in spring 2004. Seven of the ten pest mammal species present were eradicated but three species persist. Since the eradications, many native species have recovered, and several others have been reintroduced as contributions to regional and national biodiversity conservation. Public access remains unrestricted with 160,000 visitors per year. Potential pest incursion pathways include coastal ends of the fence, arrival via park activity and visitors by land or from the coast. To date, individuals of all but one of the eradicated species have been detected within the sanctuary, including an *in situ* breeding population of *Rattus rattus* in 2008. All incursions have subsequently been removed. Five years' operational experience gives a greater understanding of the incursion profile of the site and the required management responses. We remain confident that such incursions can be detected and removed without compromising existing biodiversity values and future restoration potential of the site. Although the incursion risk profile of Tawharanui Open Sanctuary is greater than most sites, lessons learnt about the detection and removal of pest incursions and surveillance management are applicable to most insular situations.

Keywords: Invasion potential, invasive mammals, pest proof fences, brushtail possum, cat, weasel, stoat, Norway rat, ship rat

INTRODUCTION

The distinction between the eradication and control of unwanted organisms is becoming increasingly blurred as technical advances increase our ability to manage riskier sites with greater reinvasion potential. In some cases, this increased operational risk may violate some of the criteria or definitions commonly used for pest management. For example, eradication is the permanent removal of a target pest species from a managed area. Several authors (e.g., Parkes 1993) describe conditions that must be met to achieve eradication as: 1) all animals can be put at risk by the eradication technique(s); 2) the animals must be put at a risk at a rate exceeding their rate of increase at all densities; and 3) immigration must be zero. This last criterion is violated by undertaking eradications in locations where there is some immigration risk. Programmes with residual immigration risk can be justified when these risks can be managed cost-effectively, suitable habitats for the native species to benefit do not exist elsewhere, as test cases for more complex operations, and to address the aspirations of communities of interest and community groups.

In theory, the incomplete removal of target species is not eradication, it is a failure (Parkes 1993). In practice, the criteria for success are less clear when there is complete removal followed by subsequent reinvasion. Here the operational failure may be one, or a combination of, lapses in biosecurity, ineffective buffering between the managed site and pest populations, or poor surveillance management. Yet the eventual outcome resembles an eradication failure. In most cases, eradication is not necessarily the desired outcome *per se*; rather it is the release from pressures exerted by unwanted organisms upon their host ecosystems.

If pest management is undertaken where the risk of reinvasion is high, eradication may only be a temporary achievement. At such locations, eradication is an ideal but the reality may be best described as maintenance at zero density. The distinction between a series of eradication operations and ongoing detection and removal of invaders is not great. The primary consideration should be confidence that the original population was eliminated and that perceived incursions are not in fact survivors.

Clear terminology is important when practitioners and stakeholders may have divergent views of the same outcome. Stakeholders, who may include political decision makers, funding agencies and affected communities, often take an absolute view of pest removal. When pest incursions are encountered these absolute views may become feelings that either the operation has failed, or that expected outcomes were communicated falsely from the outset. This in turn can translate to erosion of support for current or future operations. Such situations reflect the first of Bomford and O'Brien's (1995) desirable criteria for eradication success: that the social and economic conditions must be conducive to meeting the critical rules. Whatever terminology is used it must be aspirational and attention applied to any attendant qualifications and communication of ongoing operational risk.

The consequences of occasional pest incursions depend on the vulnerability of the species or ecosystems under threat. Ecological resilience can increase as the restoration process progresses when pests are removed, but ecological vulnerability can also increase as new threatened taxa are reintroduced. These changes increase the imperative to act against new pest incursions, while the suite of tools required to respond effectively may need to be changed or improved.

In this paper I discuss the development of incursion response theory and describe how this was applied in a fenced sanctuary that receives periodic incursions of pest mammals.

REINVASION POTENTIAL

For the purposes of this paper, an incursion is the arrival of a species without establishment, whereas an invasion is arrival followed by establishment of a breeding population (Russell *et al.* 2008).

Every site has an incursion profile, which reflects the probability of reinvasion. The incursion probability (*IP*) for a site can be expressed by the formula

$$IP = d + a - q + p + e$$

Where d = distance from nearest or most probable source population; a = assistance (e.g., sea currents, freight and transportation); q = quarantine measures implemented at either source or recipient site to detect or remove invaders;

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p = pest species characteristics (e.g., swimming or climbing capability, breeding biology); and e = environmental factors (e.g., climate, season, population pressure, mate and food availability). The relationship between, and relative weighting of, each factor is unknown. Only d and p are constants; a, q and e are variables, with the last being mostly beyond the direct influence of management.

Incursion profiles form a continuum. Open mainland reserves with sustained pest control are infinitely re-invasible at their edges and thus have IP=1. Islands that are closed by virtue of their management, legal status, remoteness, or environment can have IP near zero. However, since incursions can reach all sites, including remote islands, no site has IP=0. Between the extremes of the continuum cluster a suite of fenced peninsulas, ring fenced mainland reserves and inshore islands. Some of these may have incursion probabilities nearer to that of oceanic islands than open mainland reserves. Where these sites sit on the continuum can be heavily influenced by human activity. These anthropogenic factors also mean that IP is not constant through time, but is affected by complacency, improved knowledge, management regime change, and social pressures.

The probability of pest mammal incursion is not the sole determinant of the security of a managed site, it merely describes the risk. The biological consequences of any incursion event are determined by the managers' ability to intervene, and their confidence that new incursions can be detected.

Timeliness of detection is important in two regards. First, there is a biological imperative to detect and remove an incursion before there is unacceptable biodiversity loss, and before the incursion becomes an invasion. This is consistent with the third of Bomford & O'Brien's (1995) desirable criteria for eradications: animals surviving the eradication campaign should be detected and dealt with before an increased population becomes obvious.

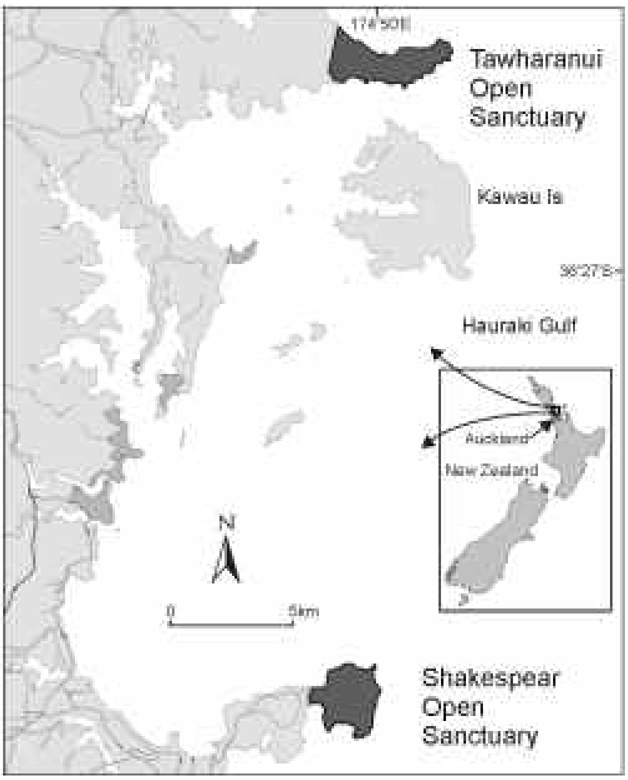

Fig. 1 Tawharanui and Shakespear Open Sanctuaries are on the east coast north of Auckland City.

The second imperative is financial. The scale of any incursion is the greatest determinant of the cost of managing such an event. Scale must be considered both spatially (area covered) and temporally (time taken to return to 'normal' management). Scale and subsequent resources and techniques to address the issue can become constrained as scale increases. Some options (e.g., aerial toxin application) may be untenable on biological (non target impacts), financial, or socio-political grounds (i.e. objections to methodology or constraints on other activities). Any cost of managing pest incursions carries an opportunity cost of other desired conservation management activity.

Detection confidence can be expressed by the formula $DC = d + r + t + p + h$

Where d= number and density of detection devices; r = reliability of devices and operators; t = time interval (exposure); p = pest species characteristics; h =habitat condition (e.g., prey and cover availability affecting pest animal ranging). The relationship between, and relative weighting of, each factor is unknown. However, the first three factors are in the manager's hands to influence.

Animals may be detected away from the point of incursion, so conclusions should not be hastily drawn regarding potential defensive weaknesses. The ranges of incursive or displaced animals can be far in excess of normal behaviour (Russell *et al*. 2005) in response to social isolation, and the animals' need to determine the presence of competitors, predators, prey and breeding opportunities.

CASE STUDY: INCURSIONS AT TAWHARANUI OPEN SANCTUARY

Tawharanui Open Sanctuary is a management layer at Tawharanui Regional Park 50km northeast of Auckland, New Zealand (Fig. 1). The park is administered by the Auckland Council in partnership with a community group: Tawharanui Open Sanctuary Society. The open sanctuary philosophy integrates the varied land uses of recreation, conservation and farming. Public access is unimpeded with approximately 160,000 visitors per year including a 260 person capacity camping ground. A 2.7km Xcluder coast to coast pest proof fence isolates 550ha of the peninsula as a barrier to the passage of mammalian pests, which enables the isolated area to be managed as a 'virtual island' (Day and MacGibbon 2007).

Mammalian pests were eradicated in spring 2004 using two aerial applications of brodifacoum (Pestoff 20R) toxic baits supported by trapping, hunting, poisoning at bait stations, and detection dogs. Ten species of pest mammals were targeted for eradication including brushtail possum (*Trichosurus vulpecula*), cat (*Felis catus*), ferret (*Mustela furo*), stoat (*M. erminea*), weasel (*M. nivalis*), ship rat (*Rattus rattus*), Norway rat (*R. norvegicus*), house mouse (*Mus musculus*), European rabbit (*Oryctolagus cuniculus cuniculus*), and European hedgehog (*Erinaceus europaeus occidentalis*). Seven of the ten species were eradicated but house mice, rabbits, and hedgehogs persisted.

In the five years following the eradications, previously absent fauna have recolonised, breeding success of resident threatened native species of flora and fauna has improved, five absent species of birds and two species of reptiles have been reintroduced, and species of fauna have been

Table 1 Animal pest incursions at Tawharanui Open Sanctuary 2005-2010

Species	n incidents	n individuals
Brushtail possum	3*	9
Cat	>50*	4*
Weasel	4	4
Stoat	2	2
Norway rat	6	10
Ship rat	6*	47*
Rat spp. (unspecified)	4	7

*minimum

translocated from this site to establish new populations.

These conservation outcomes were achieved despite incursions by all eradicated species except for ferrets (Table 1). Had rabbits and mice been eradicated, there would also have been incursions of these species around coastal ends of the pest proof fence. Footprints of both species have been detected in sand and there was also evidence of movement through Rhodamine B biomarker studies (Goldwater 2008).

There was no single proven vector or pathway for all of the animal pest incursions. Potential pathways included entry around the coastal ends of the fence, breaches of the pest fence, entry via the single automated vehicle gateway (which has no quarantine containment 'cell'), stowaways via visitors' vehicles and camping equipment, stowaways via park managers' vehicles or materials, and coastal landings either by animals swimming along coast or from boats moored offshore or hauled up on beaches.

Entry around coastal ends of the fence is the most likely source of incursions because at low tide up to 60m of beaches may be exposed beyond each fence terminus. The fence was not extended into the intertidal zone because of: 1) engineering challenges associated with storm swells and long shore sediment drift; 2) consequent maintenance costs of structure if implemented; 3) likely difficulty of obtaining planning consent due to conflict with coastal policy for coastal and foreshore structures; and 4) impeding coastal access being in conflict with primary role of the site for public recreation. Potential incursions were discouraged through a spiral 'koru' structure at each fence terminus. These structures were experimentally tested to increase interception, containment and deflection of animal pests (T. Day unpubl. data) and are used in conjunction with a trap and poison bait based animal pest management buffer designed to reduce pest mammal density. Both tools were used to reduce pest animal encounters with the ends of the fence.

Until 2008, we were confident that we could detect and remove any incursions, which had involved one or few individuals rather than populations or invasions with *in situ* breeding. The question of whether detected animals were survivors of the eradication or new incursions was addressed through the time to first capture or the time elapsed between events. Such data generally confirmed that most detected animals were new incursions. Some incursions involved multiple individuals and some individuals invaded multiple times. For many of these incursions, including those for all mustelids, the first sign of an incursion was a dead animal in traps used in the fixed surveillance network.

Once detected, each incursion triggers a management response. With incursions by a few individuals, localised response can be invoked with tools and on a scale relevant to each target species. In circumstances where toxic baiting is employed, carcasses may not be recovered to show that an animal has been killed. This absence of proof of removal can be challenging. We assume that the absence of new sign for a minimum of one month is evidence of successful interception. We do not assume that first or any capture is the last or only invader and maintain heightened surveillance for a minimum of one month after last sign detected. Throughout these responses, routine surveillance continues throughout the entire sanctuary.

In December 2007, three areas of rat activity at separate locations were detected using tracking tunnels during routine monthly surveillance. Localised response activity at the three sites resulted in captures of *Rattus rattus* at two of them. Another month of control/surveillance revealed no further sign at two sites, but the third provided further captures including juvenile rats. This evidence of *in situ* breeding resulted in a shift in response activity from localised incursion to invasion and a corresponding escalation of management activity.

Four phases of invasion response were implemented at Tawharanui: 1) detection and delimitation; 2) containment to prevent further spread; 3) eradication of animals contained in area; 4) withdrawal and review. These phases are hierarchical but can overlap. The tools and methods deployed can concurrently or sequentially serve to deliver phases 1 through 4 entirely or in part.

The process of incursion management is as important as the method employed, especially the rationale forming the basis of each management action. Attempts should always be made to follow the principles of a formal adaptive management process of model testing and refinement.

Responses to the 2007-2008 ship rat invasion at Tawharanui Open Sanctuary followed the process described above, and escalated sequentially according to information derived from the detection and delimitation phase. This was augmented by further delimitation information produced during the containment phase. Efforts focussed on the unknowns of the situation because effective management must be guided by quality information. Within reason, we could ignore the known population as long as it was contained, which allowed resources to be concentrated on implementing the incursion response.

The final area delimited in the 2007-2008 incursion was approximately 240ha, or about half the sanctuary, and was reached in four escalations. The final area was probably related to dispersal behaviour of the rats, coupled with the time it took to detect dispersing individuals. Demarcation lines need to be conservative if any statement is made as to where animals are not being supported by evidence from searching. At no time was our attention entirely focussed on the "known invasion" zone; the fixed surveillance network continued to operate with increased intensity and attention.

In order to determine the extent of invasion, all ship rat carcasses recovered (n=36) underwent genetic analysis to test levels of relatedness between individuals to determine the number of 'founding invaders'. It is assumed that due to the use of poison as well as traps that many carcasses were not recovered and could not contribute to this analysis. A pairwise relatedness estimate was used to assess the prevalence of novel or shared alleles. There were limitations to the genetic analyses because we lacked baseline information and relatedness could be imported through parents, siblings or cousins already present just outside the fence. Nonetheless, the values obtained indicated a combination of multiple founders and *in situ* breeding (D. Gleeson pers. comm.). The invading ship rat population was eradicated and the site status of zero density rats was reclaimed.

DISCUSSION

The managers' challenge is when to stand down the incursion response, i.e. how to determine 'stopping rules'. Station checks with nil positive sign provide absence of evidence rather than evidence of absence. Each check consumes resources and carries an opportunity cost for resources that may be deployed elsewhere. If effort expended on incursion response is plotted against time, the objective is to produce a steep downward trajectory. Alongside this, there should also be confidence that reduced effort will not induce unforeseen negative effects that require renewed effort not just to intercept the incursion, but also to prevent further losses of biodiversity. Thus decisions to withdraw must be inherently conservative.

Surveillance networks must detect incursions before breeding populations of pests establish, or before rare and vulnerable native species can be negatively impacted. This means that surveillance devices must be well maintained in order to avoid 'false negative' detection through malfunction, overgrowth with vegetation, or being 'swamped' with non-target activity. Similarly they need to be easily found by new staff (K. Broome pers. comm.). Such networks must also be supported by the capability to increase response efforts at short notice. Decisions are required about whether to maintain a fixed network, to keep contingency response inventory in storage, or to have some combination of both. The ongoing maintenance costs of a fixed network must be balanced against deployment costs and subsequent lost time of the stored contingency option. The network chosen needs to be easily converted from routine surveillance, to delimitation, and incursion response. The network will then need to be converted to post incursion monitoring and back to routine surveillance. These changes need to be achieved by varying the intensity and scale of checking without the need for substantial new equipment or infrastructure. The tools themselves should be adaptable to different phases, i.e. tracking tunnels for delimitation reconfigured as snap traps or bait stations for control (K. Broome pers. comm.). This adaptability addresses the resources required for deployment while overcoming potential neophobic responses from target animals.

Pest management buffering, biosecurity, surveillance, incursion response and escalation are very resource intensive. However, these are crucial to protecting the initial investment of the eradication and restoration programme and subsequent improvement in condition. If the resources do not allow for these management actions, the viability of the project becomes compromised, and the social and economic conditions are not conducive to meeting the critical rules for an eradication (Bomford and O'Brien 1995).

The 'stopping rules' must also address the possibility that the pest free state prior to incursion may not be recoverable, and that further investment of resources will not increase likelihood of achieving this. Such decisions are difficult to make as they signal the end of the dream for many stakeholders.

The Tawharanui Open Sanctuary project has demonstrated the realities of managing the aspirations of a community partnership. Significant biodiversity gains have been achieved despite considerable management challenges. An adaptive management approach has improved our management of the sanctuary and information gaps have been identified and in some cases addressed. The operational success has been sufficient to give us the confidence to undertake a similar open sanctuary at Shakespear Regional Park, Whangaparaoa Peninsula, New Zealand (Fig. 1). Here 500 ha will again be fenced to exclude mammalian pests and a suite of species similar to those at Tawharanui will be eradicated. At Shakespear, there are likely to be greater operational challenges due the proximity of 30,000 households and annual park visitation by 550,000 people.

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Invasive alien fauna in Sri Lanka: National list, impacts and regulatory framework

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Abstract In Sri Lanka, 12 invasive alien species (IAS) of animals are nationally listed,10 of which are vertebrates (seven species of fresh water fishes, two species of rodents, and one species of large mammal) and two are invertebrates (two species of molluscs). The list was created after a risk assessment based on the potential ecological and socio-economic impacts, invasive potential, distribution and the management options of the candidate species. Of the IAS where information on the year of introduction is available, four were introduced before 1978 (the year of introduction of open economic policies) and three thereafter. The main impacts of IAS on native species have been through direct destruction, competitive exclusion, and hybridisation. Four main legal enactments and three national policies are aimed at the control of entry and spread of invasive alien fauna in Sri Lanka. Despite many sectoral policies, laws and regulations touching on IAS, the regulatory framework still remains unclear, piece-meal, overlapping and largely un-enforced. A well-coordinated institutional mechanism for an effective eradication/control IAS in the country is urgently needed.

Keywords: Sri Lanka, risk assessment, predation, competition, hybridisation, eradication

INTRODUCTION

Sri Lanka is an island nation with a land area of 65,610 km^2, additional territorial waters and an Exclusive Economic Zone (EEZ) of 517,000 km^2. The country is biologically diverse, due to variations in topography and climate. Natural ecosystems and habitats include forests and grasslands, freshwater and marine wetlands, rivers, streams, mangroves, and coral reefs.

Together with the Western Ghats of India, Sri Lanka was identified by Conservation International (CI) as one of 34 global biodiversity "hotpots", with a high concentration of endemic species, and the loss of over 75% of the primary vegetation (Mittermeier *et al*. 2005). Myers *et al.* (2000) identified this region as one of eight biodiversity hotspots based on the number of endemic plants and vertebrates, their density, and remaining primary vegetation relative to the original extent. Birdlife International (BI) has identified Sri Lanka as one of the world's 356 endemic bird areas (www.birdlife.org). Sri Lanka's lowland rainforests, montane rainforests and south-western rivers and streams are listed in WWF's Global 200 eco-regions as one of the most biologically distinct terrestrial, freshwater, and marine eco-regions of the planet, and are considered priorities for conservation (www.worldwildlife.org/science/ecoregions/global200.html).

Invasive alien species (IAS) have resulted in major impacts on biodiversity at a global scale, where at least 39 per cent of the species extinctions during the past 400 years are due to IAS (www.indiaenvironmentportal.org.in/node/38152). In Sri Lanka, many alien species imported for agriculture have established in the wild in low numbers, often with few recorded effects on local ecosystems. A small proportion of intentional and accidental introductions have become serious problems that have destroyed or displaced crops or indigenous species. The contribution of IAS to habitat degradation is second only to the direct negative impact caused by humans.

Sri Lanka has now recognised IAS as a major threat to the native biodiversity (IUCN and MENR 2007). These threats have become more significant over the past two or three decades due to more liberalised economic policies facilitating international trade, travel and transportation movement (Marambe *et al.* 2003), and natural and man-made disasters supporting the free movement of international aid. The IUCN Invasive Species Specialist Group's (ISSG) Global Invasive Species Database lists 82 potentially invasive species as present on the island. More than 60 of these are known to have become invasive (40 plants and 20 animals, including 23 of "100 of the world's worst"). The rapid spread of these species in a multitude of environments makes control difficult as options applied one ecosystem may be difficult to apply in another.

Previous studies have focussed on invasive alien flora rather than on fauna (Marambe 1999, 2000, 2008) for which lists were based on limited literature, popular articles, and observations and perceptions of scientists/environmentalists. No formal risk assessment process has been undertaken to determine their invasiveness. This paper provides the most recent overview of the status and impact of invasive alien fauna in Sri Lanka, plus a review of the existing regulatory framework and strategies adopted to overcome threats from these species.

INVASIVE ALIEN FAUNA IN SRI LANKA

Bambaradeniya (2000, 2002) listed twenty species of invasive alien fauna spreading in the natural and semi natural ecosystems in different bioclimatic zones of Sri Lanka. This included nine species of freshwater fish, one of reptile, five of mammals, and five of molluscs. Ten of these species are included in the list of 100 of the world's worst IAS (IUCN-ISSG 2001). Excluding Northern and Sabaragamuwa Provinces, there are published provincial lists based on observations by scientists (Table 1), but not all of these species have been through a risk assessment process.

A risk assessment protocol for assessing the invasive alien fauna has been developed and is accepted by the Biodiversity Secretariat (BDS) of the Ministry of Environment of Sri Lanka (MESL). It evaluates invasive fauna according to stratified criteria identified under four thematic areas: potential ecological and socio economic impacts; invasive potential; distribution; and management of the candidate species (Ranwala 2010).

The national list of invasive alien fauna (Table 2) identified from this risk assessment includes seven species of freshwater fish, two species of rodents, one species

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Table 1 Distribution of invasive alien fauna in the seven provinces of Sri Lanka* P = recorded as present (Adopted from: Silva and Kurukulasuriya 2010).

Invasive Alien Fauna	NW	NC	UP	WP	CP	EP	SP
Clown knifefish (*Chitala ornata*)	P	P	-	P	-	-	P
Plecostomus catfish (*Hypostomus plecostomus*)	P	-	-	P	-	-	-
Walking catfish (*Clarias batrachus*)	P	-	-	P	-	-	-
Guppy (*Poecilia reticulata*)	P	-	-	P	-	-	P
Western mosquitofish (*Gambusia affinis*)	P	-	-	P	-	-	-
Mosambique tilapia (*Oreochromis mossambicus*)	P	P	P	P	P	-	P
Carp (*Cyprinus carpio*)	-	-	-	P	-	P	-
Snakeskin gouramy (*Trichogaster pectoralis*)**	-	-	-	-	-	-	-
Red eared slider turtle (*Trachemys scripta*)	-	P	-	P	-	-	-
House mouse (*Mus musculus*)	-	-	-	P	-	-	-
Ship rat (*Rattus rattus norvegicus*)	-	-	-	P	-	-	-
Feral cat (*Felis catus*)	-	-	-	P	-	-	-
Feral dog (*Canis familiaris*)	-	-	-	P	-	-	-
Feral buffalo (*Bubalus bubalis*)	P	-	P	-	-	-	P
Apple snail (*Pomacea diffusa*)	-	-	-	P	-	-	P
Giant African snail (*Lissachatina fulica*)	-	-	-	P	-	-	-
Field slug (*Laevicaulis alte*)	-	-	-	P	-	-	-
Rainbow trout (*Oncorhynchus mykiss*)**	-	-	-	-	-	-	-
Garden slug (*Deroceras reticulatum*)**	-	-	-	-	-	-	-
Garden slug (*Deroceras caruanae*)**	-	-	-	-	-	-	-

*NW – North Western Province, NC – North Central Province, UP – Uva Province, WP – Western Province, CP – Central Province, EP – Eastern Province, SP – Southern Province.
** These species have not been recorded in any of the above provinces despite been listed in the previous national lists (Bambaradeniya 2000, 2002; Marambe *et al*. 2001; Wijesekera and Bambaradeniya 2007)

of large mammal, and species of molluscs. In addition, 16 species have been identified as alien fauna with a potential to become invasive and eight species listed by Bambaradeniya (2002) lack recent records (see Table 1).

IMPACTS OF INVASIVE ALIEN FAUNA IN SRI LANKA

Threats posed by IAS to native species include direct exploitation or destruction, competition for resources, hybridisation and the other impacts. The following section illustrates with specific examples the likely impacts of IAS in Sri Lanka, under the above impact categories.

Direct exploitation/destruction of native species

The clown knifefish (*Chitala ornata*) is a large predator introduced in 1994. Subsequently, there have been decreases in the abundance of native fish such as *Aplochielus dayi, A. parvus, Horadandiya athukorali, P. vittatus, P. bimaculatus, R. daniconius* and *Amblypharyngodon melettinus* (Gunawardena 2002). The predatory walking catfish (*Clarias batrachus)* also has direct effects on native species (Weerawardane and Dissanayake 2005).

The guppy (*Poecilia reticulata*) was introduced to control mosquito larvae based on its larvivorous feeding habits, but its efficacy as a bio-control agent is now questionable. Research by Shirantha *et al.* (2008) showed that guppy feeding habits have become more carnivorous and the species is now feeding on the eggs of amphibians (Bambaradeniya 1999).

Feral populations of cats (*Felis catus*) and dogs (*Canis familiaris*) prey on wild reptiles, birds and small mammals (www.sundaytimes.lk/030615/funday/2.html). Feral dogs have been seen attacking wild animals in Bundala National Park (Bambaradeniya *et al*. 2002) and the dogs also avidly search for and feed on the eggs of marine turtles in coastal areas (De Silva 1999, Ilangakone 2000, Bambaradeniya *et al.* 2002). De Silva (2007) has documented domestic cats destroying herpetofauna in home gardens.

Superior competitors for resources

In Sri Lanka, Mosambique tilapia (*Oreochromis mossambicus*) is non-selective in its diet and breeds prolifically, enabling it to colonise tanks, reservoirs and slow flowing rivers while displacing native inhabitants such as *Labeo porcellus* and *L. dussumieri* (Pethiyagoda 1999). The diet of small tilapia comprises zooplankton, which are food resources for indigenous fish. The endemic red-fin labeo (*L. lankae*) overlaps in distribution with tilapia and has been driven to near extinction, possibly due to this competition (Pethiyagoda 1999, 2006).

Mozambique tilapia also occupies the same habitats as the indigenous cichlid *Etroplus suratensis*, and the two species probably compete for nesting space (Ahamed and Dharmaretnam 2008). The listing of Mozambique tilapia as an IAS was challenged by aquaculture specialists who claimed that endemic fish species do not exist in the reservoirs where tilapias are abundant (Amarasinghe *et al.* 2006). Populations of Mozambique tilapia that established in some non-flowing habitats showed little significant dietary overlap with indigenous fish species (Amarasinghe *et al.* 2008). These contradictory views indicate that the impact of co-occurring populations of tilapia and indigenous fish is not clear and further assessment is warranted.

The tank cleaner (*Hypostomus plecostomus*) can out-compete native biota. The species is an omnivore with a diet varying from plankton to plant matter and invertebrates. Further invasion to inland waters may pose a threat to endemic fish species (Wijethunga and Epa 2008). The scrape feeding habits of the tank cleaner could change habitat quality, leading to detrimental effects on co-occurring species (Amarasinghe *et al.* 2006).

In the dry zone, feral buffaloes (*Bubalus bubalis*) compete for food with herbivores such as deer (*Rusa alfredi*), sambur (*R. unicolor*) and elephants (*Elephas maximus*). Their wallowing muddies aquatic habitats, which deters their use by other animals such as elephants (Bambaradeniya 2000). In Sinharaja rainforest, exotic

Table 2 The National List of Invasive Alien Fauna and their summary status in Sri Lanka.

Species	Mode of Introduction	Spread	Nature of threat	Control
Plecostomus catfish/ Tank cleaner/ Sucker mouth catfish (*Hypostomus plecostomus*)	1994; Negligence; Ornamental fish trade	Coastal flood plain, mainly around Colombo, Gampaha, Kandy and Kalutara districts	Superior competitors for resources Scrape feeding habits-change the habitat quality	Not available
Mosambique tilapia (*Oreochromis mossambicus)*	1952; Deliberate; commercial fishery	Island wide	Superior competitors for resources	Not available
Clown knifefish (*Chitala ornata*)	1994; Neglect; Ornamental fish trade	Coastal flood plain Streams and reservoirs - wet zone	Direct exploitation or destruction of native species	Not available
Ship rat (*Rattus rattus)*	Accidental; Ships	Island wide distribution in natural and managed terrestrial habitats	Agricultural pest; hybridisation with the native biota; vector for leptospirosis virus	Chemical control – poisonous baits
Apple snail (*Pomacea diffusa*)	1980; Negligence; Ornamental fish trade	Colombo, Kalutara, Kandy, Galle, Rathnapura, Gampaha, and Matara	Destruction of aquatic plants	Not available
Guppy (*Poecilia reticulata)*	1930; Deliberate; mosquito control	Lowland wet zone, and more riverine areas - upper catchments of Mahaweli & Kelani rivers	Direct exploitation or destruction of native species	Not available
Walking catfish (*Clarias batrachus)*	Negligence; Ornamental fish trade	Marshes and streams - lowland wet zone	Direct exploitation or destruction of native species	Not available
Feral buffalo (*Bubalus bubalis)*	Deliberate; Animal husbandry	Island wide - Forests	Superior competitors for resources; hybridisation with native biota; facilitate the spread of invasive alien plants	Not available
House mouse (*Mus musculus)*	Accidental; Ships	Island wide distribution in natural and managed terrestrial habitats	Agricultural pest; hybridisation with the native biota; vector for leptospirosis virus	Chemical control – poisonous baits
Western mosquito fish (*Gambusia affinis)*	Deliberate; mosquito control	Marshes, ditches and streams of the lowland wet zone	Not known	Not available
Carp (*Cyprinus carpio*)	1915; Deliberate; commercial fishery	Headwater streams 1500m a.s.l. elevation	Superior competitors for resources; feeding habits-change the habitat quality; direct exploitation or destruction of native species	Not available
Giant African snail (*Lissachatina fulica*)	1840; Negligence, Research/Hobby	Island wide distribution in natural and managed terrestrial habitats	Pest of agricultural landscapes	Chemical control - metaldehyde

ship rats (*Rattus rattus*) appear to suppress numbers of the endemic *Srilankamys ohiensis*, which suggests competition between the two species of rats for resources (Bambaradeniya 2000).

Hybridisation with native species

Domestic buffaloes have interbred with the native wild water buffaloes (*Bubalus arnee*) to form a hybrid feral population (Bambaradeniya 2002). This has probably led to the local extinction of genetically pure populations of the wild water buffalo in locations such as the Wilpattu National Park (Deraniyagala 1964). The three sub species of ship rat (*R. rattus rattus, R. r. alexandrianus* and *R. r. rufescens*), which were accidentally introduced to Sri Lanka, have probably interbred with the two local subspecies (*R. r. kandianus* and *R. r. kelaarti)* to form hybrid populations (Bambaradeniya 2000). The extent of hybridisation in buffaloes and the rats needs to be verified by further study.

Other impacts

Some invasive alien fauna have indirect influences on native biodiversity. Feral buffalo feed on the pods of the invasive alien mesquite (*Prosopis juliflora*) and facilitate the spread of this plant in the arid zone. They also disturb natural habitats allowing the establishment of invasive alien plants such as *Lantana camara* (Bambaradeniya 2000). Ship rats spread leptospirosis virus, and feral cats and dogs are vectors of rabies (www.sundaytimes.lk/030615/funday/2.html). Increased fishery pressure and the adoption of harmful fishing practices (i.e. small-meshed gill nets) to catch exotics such as tilapia and carp (*Cyprinus carpio*) have impacted non-target species such as freshwater turtles in the dry zone reservoirs (Pethiyagoda 1999).

Table 3 The main legal instruments found in Sri Lanka to deal with invasive alien fauna.

Ordinance/Act	Intention	Applications	Problems in implementation
Fauna and Flora Protection Ordinance (No. 02 of 1937, as amended)	protection, conservation, and preservation of fauna and flora of Sri Lanka and the commercial exploitation of them	Import of any animal, spawn, eggs, or larvae of any animal can only be done under the authority of a permit [Section 37(1)]; applies to all species of animals except those domestic animals - cattle, sheep, goats, horses, asses, mules, dogs, cats, domesticated pigs and domestic fowl reared as poultry; these provisions have the same effect as if they were part of the Customs Ordinance	No provisions to deal with a species already brought in under a permit, where it has subsequently become an invasive or is likely to become invasive; this Act does not apply to plants.
Fisheries and Aquatic Resources Act (No. 02 of 1996, as amended)	to manage, regulate, conserve and develop the fisheries and aquatic resources	Minister in Charge of Fisheries and Aquatic Resources, and the Minister in Charge of Trade, can prohibit or regulate the import of fish or aquatic resources. 24 species of fishes are prohibited from being imported.	No provisions to deal with a species that has become or is likely to become an invasive in the country.
Plant Protection Act (No. 35 of 1999)	to prevent the introduction and spread of any organism injurious or harmful to plants or destructive to plants found in Sri Lanka	To prevent entry of any plant or animal that may become a pest or invasive, or potential threat to plant life. When there is reason to believe that a pest is being harboured in any premises, the D G of Agriculture can direct an inspection to ascertain the situation. The Minister of Agriculture can prohibit entry of Quarantine Pests (a pest of potential economic or environmental importance that is not yet present or present but not widely distributed and being officially controlled).	The Act does not make provisions to control an introduced species or a species with a potential to be introduced that could be harmful to animals
Marine Pollution Prevention Act (No. 35 of 2008)	to prevent, control and reduce pollution in the territorial waters.	Provisions can be used to bring in necessary regulations to control and regulate the release of ballast waters in the seas of Sri Lanka or to treat them in a specified way before releasing into the waters.	Regulations are still to be made for the implementation of the Act

EFFORTS TO OVERCOME THE THREATS OF INVASIVE ALIEN FAUNA TO THE SRI LANKAN ECOSYSTEMS

Legal Instruments

Sri Lanka is a signatory to international and regional agreements related to trade, such as World Trade Organization (WTO) Agreements, South Asian Free Trade Area (SAFTA) Agreement, and to international conventions related to IAS such as Convention on Biological Diversity (CBD), International Plant Protection Convention (IPPC), and International Convention for the Prevention of Pollution from Ships (MARPOL 73/78). Sri Lanka has also enacted many ordinances/acts to impose laws governing import of fauna and flora to the country. Key ordinances and government agencies include: 1) the BDS of the MESL, which serves as the focal point for the implementation of the CBD; 2) the Department of Agriculture (DOA) of the Ministry of Agriculture (MA) of Sri Lanka, which is the focal point for IPPC related activities; 3) the Marine Environment Protection Authority (MEPA) of the MESL is the focal point for implementation of MARPOL 73/78 Convention.

The main legal enactments that have directly assisted in eradicating and controlling the entry and spread of invasive alien fauna in Sri Lanka are given in Table 3, in the chronological order of enactment.

Legal instruments and policies

Existing legislative enactments provide considerable legal support for actions against the introduction of IAS (Table 3). However, these laws can only be used in relation to specific types of invasive species. No single enactment deals with all the different types of invasive species. Approval has now been granted to develop a new act to prevent the entry of IAS and control of those already present.

The Constitution of the Democratic Socialist Republic of Sri Lanka, states that “The state shall protect, preserve and improve the environment for the benefit of the

Table 4 National level policies directly dealing with IAS

National Policy	Implementing organisation	Relevant statements for IAS control
National Wildlife Policy of 2000	Department of Wildlife Conservation	To promote ecosystem-based management of protected areas, including the eradication of alien and invasive species, subject to thorough consideration of the environmental impacts. To regulate the importation of alien organisms, including genetically-modified organisms, so as to minimise risks to the integrity of Sri Lanka’s biodiversity
National Environmental Policy of 2003	Ministry of Environment	Environmental management systems will be encouraged to be flexible so as to adapt to changing situations (e.g., climate change, invasive species and living, genetically-modified organisms) and adopt the precautionary principle
National Agriculture Policy of 2007	Ministry of Agriculture	Strictly adhere to plant protection regulations to prevent alien weeds, insect pests and diseases from entering the country

community". This governs the activities of all state, private sector and non-governmental organisations and individuals in protecting the environment. Several government institutions have developed policy statements or working mechanisms to tackle issues related to IAS (Table 4). However, key stakeholder organisations have as yet failed to create policies related to IAS, especially those that should focus on eradication.

Action plans relevant to dealing with IAS

The BDS of the MESL, as the national authority for addressing issues related to biodiversity conservation, has taken steps to formulate a National Action Plan for the Control of IAS in protected areas, as a component of the Addendum to 'Biodiversity Conservation in Sri Lanka: a framework for action' (MENR 2007). Further, the secretariat has taken an initiative to appoint a National Experts' Committee on IAS to deal with the threats of alien invasions.

The Addendum to the Biodiversity Conservation Action Plan (BCAP) in Sri Lanka (MENR 2007) listed as high priority recommendations: 1) establish an invasive species specialist group; 2) prioritise invasive alien species including GMOs, terrestrial and aquatic species; 3) prepare a national database on IAS; 4) provide funding for research on methods to control the spread of the prioritised IAS; 5) establish a national biodiversity information management committee to implement the computerised networking and establishment of meta-data base (including invasive species); and 6) strengthen human resources, technical capacity and infrastructure of the BDS of the MENR, so as to provide capacity to coordinate and monitor a comprehensive set of biodiversity indicators and programmes (including invasive species).

The need for appropriate structures and indicators for monitoring biodiversity components and coordination of action plans is recognised as an integral part of implementing commitment to the CBD (Atapattu *et al.* 2006). There is little information about monitoring activities and evaluating success of locally organised projects. A monitoring mechanism is in place for many national and international projects. However, there is almost no evaluation of the success and failures of IAS management activities. For an effective monitoring and evaluation to take place, development and use of indicators is imperative. The Addendum to the BCAP in Sri Lanka – A Framework for Action (MENR 2007) and the relevant chapter report (Atapattu *et al.* 2006) lists indicators to be used in evaluating the impact of IAS related activities.

CONCLUSIONS

There has been a significant increase in research on specific invasive alien fauna over the past five years but there is no institution/committee assigned to oversee and coordinate research and management actions. Eradicating or managing IAS requires a coordinated strategy based on cooperation among all land managers (Marambe 2001). A National Strategy and Action Plan (NSAP) was proposed for effective management of IAS by Marambe (2001) as the existing institutional design and coordinating mechanism is insufficient or ineffective in tackling IAS issues at national and regional levels.

In Sri Lanka, the regulatory framework for IAS control remains unclear, piece-meal, overlapping and largely un-enforced, despite many sectoral policies, laws and regulations. This situation has facilitated the entry to, and spread of, IAS through new pathways created as a result of expanding international trade, tourism, and transport. Different organisations are mandated to implement policies and laws governing IAS control, planning and implementation, but at present each group addresses their own institutional concerns with little consideration for overall national priorities.

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The Rakiura Tītī Islands Restoration Project: community action to eradicate *Rattus rattus* and *Rattus exulans* for ecological restoration and cultural wellbeing

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Abstract In 2003, a non-profit group, Ka Mate Nga Kiore, was set up to oversee the restoration of four Māori-owned islands off the south coast of Stewart Island, New Zealand. The first step in the restoration was to eradicate ship rats (*Rattus rattus)* from three islands and Pacific rats (*R. exulans)* from another. The eradication was funded by the *Command Oil Spill Trustee Council* which managed the mitigation money from an oil spill off the Californian coast in 1998. The funding was coordinated via *Oikonos Ecosystem Knowledge*, a non-profit USA group primarily involved in seabird research and restoration. The project was primarily to benefit sooty shearwater (*Puffinus griseus*) and to sustain a culturally important customary harvest of their chicks by Rakiura Māori. However, like all island eradications, a wide range of other species also benefited from the removal of rats. The New Zealand Department of Conservation provided technical advice and assistance for the planning and implementation of the eradication programme. This paper describes how, with appropriate funding, community and technical support, rodent eradications can be achieved on private islands. In this case, a range of institutions and individuals joined to achieve a common goal that highlighted a significant international conservation action. We urge that more international and local-community-led restoration projects be initiated in the future.

Keywords: Ship rats, kiore, sooty shearwater restoration, muttonbirding, *Puffinus griseus,* international and local community collaboration

INTRODUCTION

Approximately 21 million sooty shearwater (*Puffinus griseus*) form breeding colonies in New Zealand (Newman *et al*. 2009), mostly (53%) on the 35 'Tītī Islands ('Muttonbird Islands) around Rakiura (Stewart Island) in southern New Zealand (Fig. 1). The indigenous people of southern New Zealand are Rakiura Māori, who own these islands and have a legal right to harvest the near-fledgling chicks, which they call 'tītī' or 'muttonbirds'. Tītī harvesting is a fundamental part of being Rakiura Māori (Moller *et al*. 2009), an important source of income (Wilson 1979), spiritual inspiration (Lyver and Moller 2010) for the birding families, and a nationally important example of *kaitiakitanga* (Māori conservation management) and environmental co-management in action (Moller *et al*. 2000; Stevens 2006). Sustaining the abundance of sooty shearwaters is therefore a fundamentally important goal of the Rakiura Māori community.

On 26 September 1998, the tanker vessel "*Command*" released approximately 3000 gallons (11,356 litres) of oil off the California coast (Anon. 2004). Thousands of seabirds were killed by the spill, including between 2 and 32 thousand (median estimate 15,500) sooty shearwaters (Moller *et al*. 2003). One of eleven sooty shearwaters recovered on beaches during the spill had been banded by an Otago University research team on Whenua Hou/ Codfish Island off the north west coast of Rakiura (Stewart Island). This individual provided the required nexus to allow for mitigation funds to recover damaged natural resources under a consent decree signed by the guilty party and the US multi agency Command Spill Trustee Council. The banding programme was part of *Kia Mau Te Tītī Mo Ake Tōnu Atu* / "Keep the Tītī forever", a 14-year study into the productivity of the species and the sustainability of the muttonbird harvest (Moller 1996; Moller *et al*. 2009).

Oikonos Ecosystem Knowledge, an American non-profit research group, recognised this event as an unprecedented opportunity for *Command* mitigation funds to repair the oil spill injury to sooty shearwater populations in New Zealand. The eradication of introduced predators on New Zealand islands containing colonies of sooty shearwaters was considered the most effective way to repair the oil spill injury and also provide substantial additional multi-species benefits.

This paper describes how the funds from the oil spill, with community and technical support, enabled rodent eradications to be achieved on private islands. We also outline how institutions and individuals collaborated to achieve a significant international conservation action.

STUDY SITES

Four islands were chosen as a priority for rodent eradication, based on their importance for birding (the taking of muttonbirds)(Newman *et al*. 2008, 2009), historical significance, conservation potential, and the feasibility and cost effectiveness for predator eradication. These were Taukihepa / Big south Cape (939 ha), Rerewhakaupoko/ Solomon (30 ha), Pukeweka (3 ha), and Mokonui / Big Moggy (86 ha) (Fig. 1).

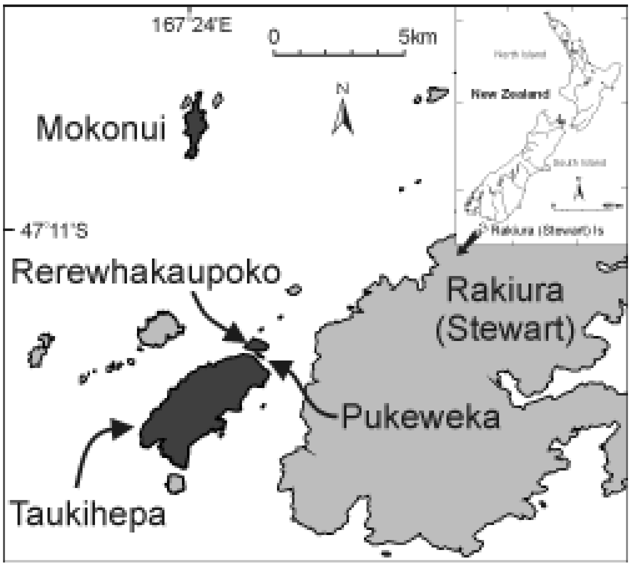

Fig. 1 The Tītī Islands, where the Rakiura Restoration Project research and rat eradication took place in 2006.

Pages 451-454 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

The Taukihepa group (Taukihepa, Pukeweka and Rerewhakaupoko) had been historically recognised as one of New Zealand's ecological jewels as the last refuge for several species of birds and the greater short-tailed bat (*Mystacina robusta*) before ship rats (*Rattus rattus*) invaded the group in 1963. The rats caused extinction of Stead's bush wren (*Xenicus longipes variabilis*) and Stewart Island snipe (*Coenocorypha aucklandica iredalei*), and perhaps also the greater short-tailed bat, and potentially the local extinction of an unknown number of species of birds, lizards, and invertebrates (Bell 1978; Ramsay 1978). It is particularly poignant that the Rakiura Restoration Project targeted rats on the Taukihepa group because it was the 1964 rat irruption and ensuing ecological disaster - more than any other event in New Zealand - that triggered widespread realisation of the ecological impacts of introduced rodents and the need for their eradication from islands (Dingwall et al. 1978).These three islands were effectively treated as one landmass during the eradication because the rat populations can easily swim between them.

The eradication of Pacific rats (kiore: *Rattus exulans*) from Mokonui, which is approximately 5 km to the west of Taukihepa, was included in the project during the early stages of planning at the request of its beneficial owners. This extension imposed only a minimal increase in planning and implementation costs, yet promised significant ecological gains because of its relatively large size.

THE PROJECT

Funding

The bid to eradicate rats from the Tītī Islands was prepared by scientists assisting the joint Oikonos-Rakiura Tītī Islands Administering Body (Moller *et al*. 2003). This successful bid to the *Command* Trustee Council provided US$513,000 for restoration including: rat eradication (70% of expenditure); scientific monitoring of outcomes (10%); reporting and administration (10%); educational video about the project (5%); and initiating community-level quarantine programmes after the rats were removed (4%).

Community Involvement

The Tītī Islands are managed under two different management committees, membership of which is based upon the history of each island. Once eradication funding had been secured, in order to facilitate the two committees working together, and effectively to provide a sub-committee which could focus on the eradication, a NZ non-profit incorporated society was formed. This group could act on behalf of the islands' owners, communicate independently with Oikonos and the Command Trustee Council, and feed back to the committees as required. The community called this group *Kā Mate Ngā Kiore* (KMNK), which loosely translated means "death to the rats". KMNK's main tasks were to: 1) link the various parties involved in the planning and operational aspects of the project with the thousands of owners of the islands; 2) keep all parties informed of progress; and 3) get a consensus on approvals from the owners for relevant actions when required. KMNK also coordinated the involvement of birders in the operational aspects of the project, which were guided by New Zealand's Department of Conservation (DOC).

Understandably, some of the American public opposed the transfer of reparation funds to New Zealand. However, the Tītī project was seen by the Trustee Council as an important part of mitigating the impact of the oil spill. The *Command Trustee Council* had confidence to support investment outside the USA because: 1) a comprehensive ecological research programme had already developed methods and collected some of pre-eradication baseline data, which built confidence in adequate documentation of repair to the oil spill injury; and 2) a research team (*Kia Mau Te Tītī Mo Ake Tōnu Atu*) had population parameter estimates on hand to demonstrate the size of the injury to sooty shearwaters and to simulate prospects for recovery.

Accountability and security of funding streams was paramount. One of KMNK's roles was to financially manage the project within New Zealand, contracting in assistance as required and ensuring that the required reporting was completed. *Oikonos* was actively involved in project management and became the liaison between USA and New Zealand entities. Effectively, a trusted local US agent oversaw funding, while the KMNK performed a similar and crucial role in New Zealand for operations and community involvement.

Planning the eradication

Planning for the eradication started in 2003 when KMNK obtained the final mandate from the islands' owners to make any decisions required to carry out the eradication. This was crucial as it was impractical to go back to all the owners every time a decision was required. In 2004, a Memorandum of Understanding (MOU) was drawn up between DOC and KMNK so that the roles and responsibilities of the two groups concerning preparation for the eradication were clearly defined (DOC 2004). The MOU recognised DOC's international expertise in rodent eradications. Technically, the eradication was considered by DOC to be relatively straightforward. However, the large number of owners of the islands, and the fact that the islands are inhabited for up to two and a half months a year, added novel complications. The trust and guidance of KMNK therefore became fundamentally important for the success of this project. KMNK also ensured that all cultural and spiritual concerns were considered. These included: 1) a blessing ceremony prior to the eradication to keep the operators safe and ask for overall success of the venture; and 2) ensuring that ancestral guardians of the islands understood the need to break a traditional *rāhui* (prohibition) that normally bans all muttonbirders from visiting the islands except during the late fledging stage. The *rāhui* protects habitat and minimises disturbance to the adults' breeding attempts (Moller and Lyver 2010).

The eradication was originally planned for the winter of 2005. However, planning and financial hold-ups delayed the operation for a year. KMNK and the *Command Trustees* agreed that it was important to not rush the eradication operation. In 2006, a contract for service was signed by DOC and KMNK for the bait drop (DOC 2006a). This replaced the MOU and detailed the roles of the two parties for the eradication itself. We believe that clear MOUs between community representatives and government agencies or researchers are essential to allow co-ordination of diverse contributions, all of which are needed for the success of the overall endeavour. In general, investment of time and resources to allow extensive communication between stakeholders slows the process down, but the multi-stakeholder buy-in to the overall goal is thereby more solid and lasting. Local knowledge of the community was also essential for putting the eradication plan into action. DOC prepared the applications for all the legal consents required, although they were applied for and issued to KMNK. This simplified the consultation process because KMNK had direct contacts with most of the affected parties and were in a better position to convince them of the benefits of the project, whereas DOC had the legal and technical experience required to obtain the consents for the release of poison bait into the environment. A significant concern for New Zealand public opposition to aerial poison baiting was addressed by having DOC manage the overall consents process.

Operational work

A detailed operational plan was developed by DOC in consultation with KMNK to ensure that all details were covered and everybody knew their roles when bait was being spread (DOC 2006b). The bait was 10 mm diameter cereal bait pellets (Pestoff 20R) containing 20 ppm brodifacoum in 25 kg bags loaded into 1.2 m^3 plywood "pods" used previously on Campbell Island (McClelland 2011). The pods were loaded on to a local charter vessel and transported to Taukihepa where they were unloaded by helicopter and placed in covered rows at a sheltered site. To ensure that pods remained water tight, their condition was monitored by an experienced contractor who was accompanied by muttonbirders from the island. The pods were flown to a preselected open location near the top of the island on the day of the bait drop. The bait loading team consisted of DOC staff, experienced contractors and volunteer local birders, with a dedicated site manager to oversee loading and safety.

The eradication followed the standard procedures developed in New Zealand over the proceeding 20 years: two aerial drops of 8 kg ha^{-1} and then 4 kg ha^{-1} (e.g., Broome 2009). Helicopters carrying underslung spreader buckets spread bait in an 80 m wide swath. Overlapping dispersal (50% for the first drop and 25% for the second) minimised the chances of gaps and two additional swaths were spread around the coast as this is recognised as a habitat typically with increased densities of rats (Taylor and Thomas 1989).

Ground baiting

More than 100 buildings are distributed around the islands, primarily near the coast. These include sleeping quarters, workhouses, and storage sheds used during the muttonbirding season. Bait was spread by helicopter over each entire island, including over buildings. However, buildings could still have provided refuges for the rats where they could obtain shelter and food and not be exposed to the bait. KMNK coordinated approximately 40 volunteer birders to go to the island on the day of the first drop and place bait in aluminium dishes in cavities within all buildings. This was a major undertaking and could not have been coordinated without local knowledge and approvals for entry into the buildings.

All water collection systems on the buildings had been disconnected during the previous birding season. After sufficient rain had fallen to clear any bait off roofs, KMNK then arranged for a team of birders to return to the island in November to reconnect the water systems so that tanks were replenished with drinking water by the time the community returned next March for the 2007 birding season.

Public outreach

As the project was recognised as being nationally significant, KMNK worked with the media, papers and television, to get coverage whenever possible. A video, recording the whole project, was produced by South Coast Productions and KMNK to highlight the cultural significance of the project as well as its technical aspects (Asher 2007). Oikonos provided updated information via The Rakiura Tītī Restoration Project webpage (http://www.oikonos.org/projects/titi.htm).

Outcome monitoring

Informal post-eradication rat monitoring was carried out by the birders, who are active around the island during both day and night for up to 75 days of the year while harvesting the muttonbirds (McKechnie *et al*. 2010). The many buildings should also have acted as attractants for any remaining rats hence, aiding in their detection. Although the monitoring was extensive, it was not formalised, there was no training, and no attempt was made to record where people had been, so there could potentially have been gaps in the coverage. We therefore waited for three years (three muttonbirding seasons) without rat sign before declaring the operation a success in June 2009. There was still no sign of rats during the March–May 2010 birding season.

The funding agency required any repair to the impacted population to be quantified. Monitoring plots were established so that a 'Before-After-Control-Impact' design (Stewart-Oaten *et al*. 1986) can eventually be used to assess to what extent rat eradication triggers increased sooty shearwater abundance. However, the median age at first breeding of sooty shearwaters is approximately 7.8 years (Fletcher *et al*. subm.), so it will be at least 2014 before initial effects of the eradication on recruitment can be detected.

Monitoring of other species has been opportunistic. The removal of the rats has allowed the recovery of terrestrial bird species including Stewart Island robin (*Petroica australis rakiura*) and fernbirds (*Bowdleria punctata*), which naturally re-established from neighbouring predator free islands. However the ongoing presence of weka (*Galliralus australis*), a large predatory rail that was introduced to the island in the early 1900s as a food source, has hindered recovery of smaller ground nesting birds, burrowing seabirds, lizards, and larger invertebrates. KMNK would like to remove weka from the islands, but currently lack the resources to do so.

Biosecurity programmes

Ongoing ecosystem and threatened species recovery depends on heightened biosecurity now the eradication is complete. Each March and April, a wide variety of vessels transfer large quantities of food-stuffs and equipment to the islands. No formal quarantine programmes existed before the eradication project. The *Command Trustee Council* and KMNK team were anxious to lock-in the benefits of the rat eradication by minimising the chances of rats re-invading by accidental transport to the islands.

New quarantine measures are focused primarily at pre-departure points and in transit because catching rodents once they reach the islands is considered unlikely. Measures include producing and disseminating posters, calendars, and other 'promotional' material all emphasising the importance of quarantine: giving presentations at 'permit' days (important pre-season administrative meetings for muttonbirders); a short film about the eradication itself, including the importance of quarantine has been produced by KMNK.

DISCUSSION

This project involved a diverse range of organisations and groups, which shows that adequate funding and the right technical advice enables private groups to carry out eradications on their own land. Direct involvement and community "ownership" of environmental management is seen as key in building 'environmentality' (Agrawal 2005) and commitment to 'Adaptive Co-management' (Berkes and Turner 2006) for long-term restoration and sustainable use of wildlife (Stephenson and Moller 2009).

The project could not have been carried out by any one of these groups without assistance from the others. *Oikonos* initiated the project and had the required understanding of the American mitigation process to convince the *Command Trustee Council* that the project was worth funding; Otago University had banded the bird that proved the vital link to the funding in the first place and had the ability to carry out the research required by the funders; DOC had the required expertise to plan and carry out the eradication; KMNK drove the whole project and co-ordinated the community of island owners. KMNK were given DOC's Conservation award in 2007 for the effective manner in which they performed this crucial role to make the project a success.

KMNK are now working with DOC to reintroduce some species of birds which were previously present on the islands. Tīeke / South Island saddlebacks (*Philesturnus carunculatus carunculatus*) will be reintroduced to Taukihepa in March 2010. The return of this sub species is especially significant as they were saved from extinction after rats invaded Taukihepa by the transfer of 36 individuals to two nearby islands (Atkinson and Bell 1973; Bell 1978). Having charismatic and culturally important species such as tīeke on the island for the first time in over a generation, should emphasise to the birders the ecological impact the rats had and encourage the owners to maintain the quarantine standards required to keep rodents off the islands.

CONCLUSIONS

The eradication of rats from the Taukihepa group is a locally and internationally significant conservation event, brought to completion by private landowners, a NZ government department, a university and a US-based international non-profit working together. Participation in the restoration project, and the goal to get rid of the rats, has been enormously appreciated by the muttonbirding community. The project is also the first time that mitigation money from an oil spill off the American coast has been spent away from the USA. This sets an important precedent in recognising that negative environmental events, such as oil spills, in one part of the world can have significant impacts on another nation many thousands of kilometres away. Agencies and countries need to work together to get the best possible results for the available money and recognise that the movements of seabirds across political boundaries and jurisdictions are ultimately irrelevant from an ecological point of view (MacLeod *et al*. 2008; Nevins *et al*. 2009).

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Eradication of stoats (*Mustela erminea*) from Secretary Island, New Zealand

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Abstract Stoats (*Mustelia erminea*) are known to be good swimmers. Following their liberation into New Zealand, stoats reached many of the remote coastal islands of Fiordland after six years. Stoats probably reached Secretary Island (8140 ha) in the late 1800s. Red deer (*Cervus elaphus*) are the only other mammalian pest present on Secretary Island; surprisingly, rodents have never established. The significant ecological values of Secretary Island have made it an ideal target for restoration. The eradication of stoats from Secretary Island commenced in 2005. Nine-hundred-and-forty-five stoat trap tunnels, each containing two kill traps, were laid out along tracks at a density of 1 tunnel per 8.6 ha. Traps were also put in place on the adjacent mainland and stepping-stone islands to reduce the probability of recolonisation. Pre-baiting was undertaken twice, first in June and then in early July 2005. In late July, the traps were baited, set and cleared twice over 10 days. Ninety-five stoats were captured in this period. Subsequent trap checks have taken place three times each year: in November, February and between May-July. Forty-four stoats were caught in February 2006, with successive captures decreasing to between 0 and 9 each trapping period, with most caught in autumn. Genetic analysis of stoats captured to June 2008 indicates that these stoats were a mixture of residents and a few immigrants. A significant stoat plague event during summer 2006-2007 may have increased the likelihood of new stoats subsequently arriving on Secretary Island. While eradication has not yet been achieved, many of our conservation objectives are being met. The experimental nature of this programme has opened the door for testing new ground in the field of island eradications and challenging some of the previously held views of what should and should not be attempted.

Keywords: Eradication, immigration, trapping, monitoring, genetic analyses, restoration

INTRODUCTION

Stoats (*Mustela erminea*) were first introduced into mainland New Zealand in the late 1880s in response to feral rabbit (*Oryctolagus cuniculus*) plagues that were destroying pasturelands and posing a serious threat to the New Zealand economy. Stoats have had dramatic effects on New Zealand's naïve native animal species, many of which evolved without terrestrial predators (King 1984). Stoats are very mobile and are capable swimmers (Taylor and Tilley 1984) and were observed by Richard Henry, curator on Resolution Island in Fiordland, by 1900 (Hill and Hill 1987). The stoats probably invaded other remote islands in Fiordland, including Secretary Island, at around the same time.

Secretary Island is administered by the New Zealand Department of Conservation. In 2004, it became the focus of a 10-year programme to eradicate stoats and red deer (*Cervus elaphus*) (Edge *et al.* 2011). Techniques for eradicating stoats from islands had been piloted successfully on several smaller islands in Fiordland (Elliott *et al*. 2010). Stoats had also been trapped on 19 islands ranging in size from 1 - 67 ha and within varying distances from the mainland over a four-year period, in order to produce a predictive model of stoat reinvasion; of 46 stoats captured, only one was caught on an island further than 304 m offshore. Based on these results, Elliott *et al*. (2010) concluded that large islands like Secretary and Resolution would be suitable for stoat eradication attempts. Our paper describes an approach to the eradication of stoats from Secretary Island based on an operational plan with two key objectives (Golding *et al.* 2005):

1. To eradicate stoats from Secretary Island. The plan defined 'eradication' as the complete removal of the resident stoat population and the establishment of a long-term control and monitoring programme to manage re-invasion.

2. To achieve and maintain a zero-density stoat population on Secretary Island so that indigenous species currently existing on the island or introduced to the island can thrive.

The scale and experimental nature of this programme required that, in addition to determining whether the outcome of these objectives is met, opportunities for learning must also be undertaken. Stoats captured after the initial eradication campaign could be used to determine the level of subsequent reinvasion to Secretary Island. Here we present all trapping data from 2005 to December 2009 and a preliminary analysis of stoat reinvasion using molecular genetic techniques described by Gleeson *et al.* (2010). A demographic study of the Secretary Island stoat population prior to trapping and after the initial knock-down is underway (A. Veale unpubl. data). Diet analysis on the original trapped population has been undertaken but is not included in this paper (E. Murphy pers. comm.).

METHODS

Study area

Secretary Island (8140 ha; 1196 m), at the entrance to Doubtful Sound on the western coastline of Fiordland National Park (Fig. 1), is the second largest island on the Fiordland coast and the third highest island in New Zealand. The island is separated from the mainland portion of Fiordland by Thompson Sound to the east (minimum distance between the two is C. 950 m), and by Doubtful Sound to the south (minimum distance to closest stoat population is two passages C. 170 m and 600 m via Bauza Island which is largely stoat-free).

In 1963, Secretary Island was designated a 'Special Area' within Fiordland National Park by the New Zealand Government due to the island's unmodified vegetation and the real (or apparent) absence of introduced browsing or grazing animals including the brushtail possum (*Trichosurus vulpecula* and red deer. Introduced rodents were also absent, making Secretary Island the largest inshore island in New Zealand free of such pests. Stoats were the only mammalian pests known to be present. Red deer probably arrived in the late 1950s (Mark and Baylis 1975), but it took some time for a population to establish (Crouchley *et al*. 2011).

Pages 455-460 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

Fig. 1 Location of Secretary Island within Fiordland National Park.

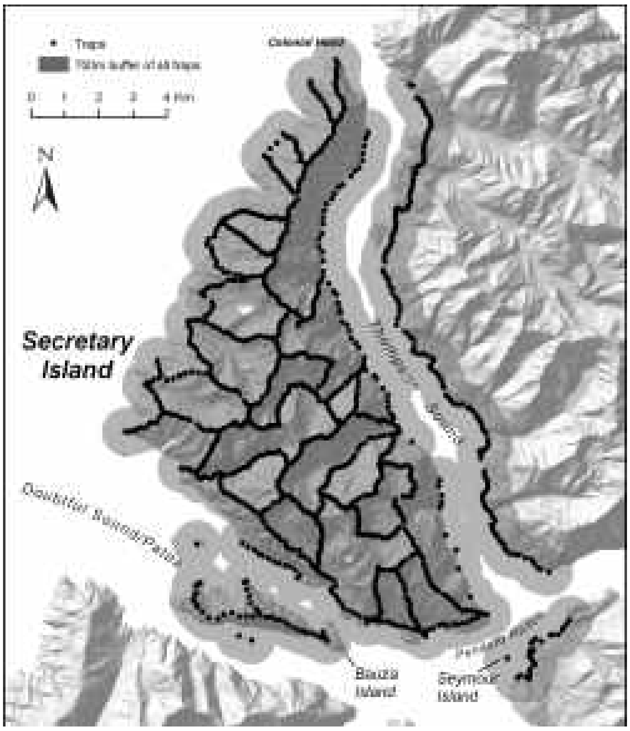

Fig. 2 Secretary Island trap lines and mainland traps buffered to 700m.

Stoat trapping on Secretary Island

Full details of methods for the trapping programme on Secretary Island are provided in the Operational Plan (Golding *et al*. 2005). In brief, these involved the following techniques.

A 108 km network of trap lines was established on Secretary Island from October 2004 to April 2005 along main ridge lines and spurs, habitat boundaries, waterways and traversable terrain (Fig. 2). Based on previous successful eradications of stoats from islands in Fiordland, we needed a minimum average density of one trap tunnel per 9 ha (Elliott *et al.* 2010.). We also needed to ensure that every stoat on the island would encounter a trap (Parkes 1990; Parkes *et al*. 2002). Home range estimates for stoats vary according to gender, season, and food availability, so it was important to determine the smallest likely home range in order to decide the maximum spacing between traps. Home ranges are smallest when prey, especially rodents, is easily available. For example, average home ranges were 93 (SE±7) ha for four male stoats and 69 (SE±8) ha for five female stoats in a Fiordland beech (*Nothofagus* sp.) forest when rodents were abundant (Murphy and Dowding 1995). Larger stoat home ranges were reported in areas where rodents are scarce, with estimates of 204 ha for males and 124 ha for females (Murphy and Dowding 1995), 223 ha for males and 94 ha females (Alterio 1998), and 210 ha for males and 89 ha females (Miller *et al*. 2001). Without similar home range information for stoats on Secretary Island, our trap network was based on the smaller home range sizes of Murphy and Dowding (1995), which meant that wherever possible traps should be no more than 700 m apart. However, due to the extremely rugged terrain on the island there were seven locations where this distance exceeded 700 m (see Fig. 2).

A total of 945 tunnels each containing 2 Mark IV Fenn kill traps (DB Springs Ltd. Worcestershire, England) were placed at 135 m intervals along the trap lines and at 150 m intervals along the eastern coastline (accessible by boat), yielding an average tunnel density of 1 tunnel per 8.6 ha (Fig. 2).

Two tunnel types were used to house traps: 300 wire mesh tunnels with wooden bases and 645 wooden tunnels with wire mesh ends. Wooden and wire mesh tunnels were placed in a repeated sequence along trap lines comprising one wire tunnel followed by two of wood. The variation in tunnel types was used to overcome any possibility that a few stoats were unwilling to enter either one type of tunnel.

Previous stoat eradications in Fiordland used pre-baiting whereby stoats were free to enter traps set with the safety catch on and with bait was left inside and outside the trap entrance. It is not possible to determine how crucial pre-baiting has been to the success of these programmes. Pre-baiting is relatively inexpensive and the amount of bait-take observed during the pre-baiting phase suggests it may reduce the time taken to achieve the initial knock-down. Traps were pre-baited twice on Secretary Island: 20 June - 26 June and 5 July - 11 July 2005. During pre-baiting, each tunnel site was baited with one fresh hen's egg and a piece of meat (ca 3 cm cube of beef, rabbit or venison) on the bait block between traps. An additional hen's egg was also placed outside the trap tunnel on the ground and another approximately 1 m off the ground on a tree.

Stoat trapping began on Secretary Island from 20 - 30 July 2005 using the pre-baiting regime. Traps were checked twice during this initial trapping period and were only re-set or re-baited if required. Thereafter traps were left set and baited, then serviced three times annually in November, February and between May and July.

The location, tunnel type, type of bait used, weight and sex of each stoat trapped were recorded and the carcass frozen for future analyses of diet and aging using cementum analysis of teeth. Sex was determined from the presence/absence of a baculum bone, unless the specimen was badly degraded, in which case it was recorded as "unknown". Tissue or bone samples were taken from all stoats captured for DNA analysis.

In July 2006, all wire mesh tunnels were removed from the island due to disturbance from native birds such as kaka (*Nestor meridionalis*), kea (*N. notabilis*), and weka (*Gallirallus a. australis*). Concurrently, each wooden tunnel was modified to contain a single trap. In July 2007, all remaining traps were replaced with single-set stainless steel DOC 150 traps (CMI Springs Ltd. Wellington, NZ).

Managing reinvasion

A coastal trap line comprising 180 double-set DOC 150 traps in wooden tunnels was established on the mainland along Thompson Sound and Pendulo Reach (Fig. 2). Stepping-stone islands to the south and south-east were already being trapped as part of the Fiordland Stoat Immigration Study (Elliott *et al*. 2010). Trapping on the mainland (hereafter referred to as Mainland) commenced in March 2005 with a subsequent check during the initial knockdown on Secretary Island. Thereafter, traps were serviced in November and February, which activated traps with fresh bait immediately before juveniles left their natal den, and cleared the traps after most juveniles had dispersed to establish new territories (King and Powell 2007).

Monitoring for stoats at low density

Tracking tunnels were not used to monitor stoat activity on the island because their rigorous use required a very large number to be set and serviced (Brown and Miller 1998; Choquenot *et al.* 2001; King *et al.* 2007).Given that the probability of a stoat entering a tracking tunnel and a kill-trap tunnel is similar, we viewed dead stoats in traps as preferable to stoat footprints in tracking tunnels. Based on the success of previous stoat eradication operations in Fiordland up to 2004, we assumed that kill-traps would provide good detectability for stoats at low density. Trained stoat-indicator dogs were used on and off the tracks and we also requested contract deer hunters to record their observations of stoat sign.

Molecular data

Molecular DNA techniques have successfully identified survivors from invaders following island rat eradication programmes (Adbelkrim *et al.* 2007; Rollins *et al.* 2006; Russell *et al*. 2010). To be useful, the technique requires measurable genetic differentiation between sample populations. We used molecular analysis to determine the frequency of immigration by stoats to Secretary Island following the initial knockdown.

DNA extraction and microsatellite amplification

Of 189 stoats caught, 89 were used in the genetic analysis. Fifty-four stoats were from Secretary Island, including 10 from July 2005, 25 from February 2006, 5 from February 2007, 6 from May 2007, 1 from June 2007, 1 from January 2008, and 7 from June 2008. Thirty-four samples were obtained from stoats trapped on the adjacent Mainland from July 2005 - January 2008. Sub-sampling from July 2005 and February 2006 was random, thereafter all of the stoats captured were analysed for each of the stated time periods. The intention is to include all of the stoats caught for all time periods for future analysis.

Tail tissue samples were dissected in the laboratory, where 50 mg of muscle tissue and caudal skin were removed. DNA was then isolated, using a Bio-Rad AquaPure Genomic Tissue Kit (Cat# 732-6343) following the manufacturer's protocol, and re-suspended in 100 µl of supplied buffer.

All samples were genotyped using sixteen microsatellite loci developed from a range of mustelid species. Primers used were MER005, MER030, MER022, MER041, MER009, and MER082 developed from *M. erminea* (Fleming *et al.* 1999); MVI057 developed from *M. vison* (O'Connell *et al.* 1996); WE7 and WE8 from *M. sibirica* (Huang *et al.* 2007); MLUT27 and MLUT32 developed from *M. lutropola* (Cabria *et al*. 2007); MA1 developed from *Martes americana* (Davis and Strobeck 1998); MEL1 and MEL4 developed from *Meles meles* (Bijlsma *et al*. 2000); RIO11 and RIO19 developed from *Lantra canadensis* (Beheler *et al.* 2005). PCR amplification and genotyping followed Gleeson *et al.* (2010).

DNA analysis

For statistical purposes, the data were grouped into three 'populations': 1) Secretary Island residents (n=35) consisting of 10 from the initial knockdown and 25 trapped in February 2006 (these latter were mostly juveniles and considered to be survivors from the initial knockdown); 2) all stoats trapped from February 2007 – June 2008 (n=20); and 3) all samples from the nearby mainland site (n=35) from July 2005 - January 2008.

Microsatellite genotypes were analysed using GenALEx v. 6.2 (Peakall and Smouse 2006) to generate observed and expected heterozygosities, allele frequency scores and Hardy-Weinberg equilibriums. Pairwise F_{ST} parameters for each population pair were estimated according to Weir and Cockerham (1984). The data were analysed using the Bayesian clustering method implemented in STRUCTURE ver 2.3 (Pritchard *et al.* 2000) to provide another estimate of pairwise F_{ST} parameters and to determine the number of distinct genetic units (K) in the dataset. This method does not require prior knowledge of sampling localities and assigns individuals into groups minimising deviations from Hardy-Weinberg proportions and genotypic linkage equilibrium. The admixture model with correlated allele frequencies was chosen. Ten replicates were conducted for each run, consisting of a burn-in period of 100,000 MCMC (Markov Chain Monte Carlo) steps followed by 10^6 iterations. The ΔK method of Evanno *et al*. (2005) was applied and plots of the log posterior probability of the data [ln$P(D)$] for each value of K examined.

Assignment tests were carried out to determine the most probable origin of the individuals captured after the initial eradication operation using GENECLASS 2.0 (Piry *et al.* 2004). The likelihood of the multilocus genotype of each individual being assigned to the resident Secretary Island population or the Mainland population was calculated in order to identify putative residual individuals or migrants. Ten thousand MCMC simulations per population were run using the L_h/L_{max} likelihood computation (Paetkau *et al.* 2004). An individual was considered to be a disperser if the L_h/L_{max} P value was below 0.01.

Table 1 Sex of stoats caught on Secretary Island between July 2005 and December 2009.

Time period	Male	Female	Unknown
July 2005	34	56	5
Nov 2005	2	5	2
Feb 2006	13	28	3
July 2006	0	1	5
Nov 2006	0	0	0
Feb 2007	3	3	0
May 2007	2	3	1
Nov 2007	1	0	0
Feb 2008	0	2	0
June 2008	3	4	1
Dec 2008	0	0	0
Feb 2009	1	1	7
May 2009	0	3	1
Dec 2009	0	0	0

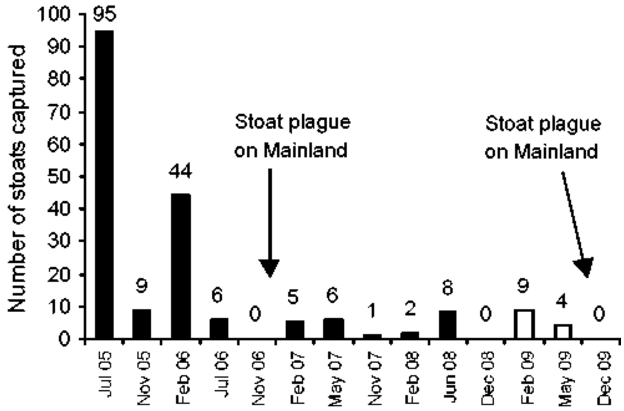

Fig. 3 Stoat captures on Secretary Island from July 2005 to December 2009. Solid bars denote the period from which trapped animals were analysed using molecular DNA techniques. Arrows indicate stoat plague events on the adjacent mainland driven by beech (*Nothofagus* sp.) masting events in the preceding autumn causing an increase in rodent numbers.

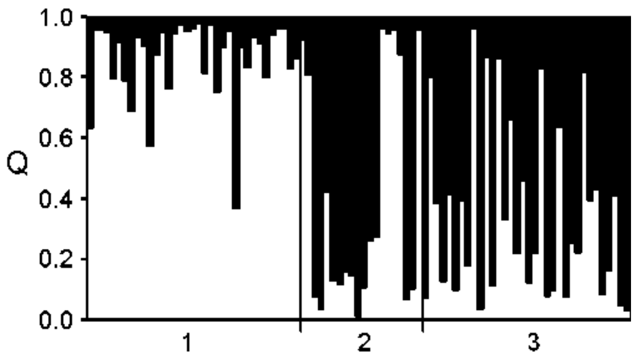

Fig. 4 STRUCTURE bar plot of estimation of the membership coefficient (Q) for each individual stoat for the three groups for K = 2. Each individual is represented by a thin vertical line, showing degree of admixture. Black lines separate individuals from each of different population groups based that are labelled below the figure.

Table 2 Summary statistics for stoats from Secretary Island and Mainland. N = Sample Size; N_A = mean number of alleles per locus; N_{PA} = number of private alleles with frequency > 0.05; H_O = observed heterozygosity; H_E = expected heterozygosity.

Location	Year	N	N_A	N_{PA}	H_O	H_E
Secretary I.	2005-06	35	4.06	0	0.498	0.539
Secretary I. post eradication	2007-08	20	4.69	5	0.491	0.579
Mainland	2005-08	34	5.06	2	0.471	0.572

RESULTS

Secretary Island stoat captures

Prebaiting and Trapping

Bait was taken from 95% and 99% of all trap tunnels during the first and second pre-baiting periods respectively. Following the knockdown in July 2005 <10 stoats have been caught in each trapping period (Fig. 3) mostly in autumn and early winter. The sex ratio of captures was approximately 2 females for every male (Table 1). Stoat captures were generally well spread across the island with highest numbers in the west and north.

Molecular analysis

No significant linkage disequilibrium was detected between loci, so all loci included in the analysis were considered independent. The mean number of alleles per population (Table 2) ranged from 4.06 for the original Secretary Island population, 4.69 for the post-eradication Secretary Island population, through to 5.06 for the nearby Mainland population. There were no alleles of frequency > 0.05 restricted only in the original Secretary Island population, while there were five alleles found only in the post-eradication Secretary Island population, and two alleles restricted to the Mainland population. Ten alleles shared between post-eradication Secretary Island and the Mainland that were not found in the original Secretary Island population.

F_{ST} values between populations were relatively low, indicating little population structuring. Pairwise estimates were lowest between the post-eradication population and the mainland (0.006), and highest between the original Secretary Island population and the mainland (0.03). The STRUCTURE analysis showed only slight differences between average loglikelihood estimates across different population scenarios ranging from K=1 to K=5. The best scenario revealed from plotting these estimates was K=2. The proportion of membership (*q*) of each group to the two inferred clusters (Secretary Island vs Mainland) (Fig. 4) shows group 2 (post-eradication Secretary Island) individuals being an admixture of both sources.

GENECLASS identified four individuals from the post-eradication Secretary Island population as first-generation immigrants from the mainland, while three individuals were assigned to the original Secretary Island population (L_h/L_{max} P < 0.01).

DISCUSSION

Unlike previous eradications on smaller islands in Fiordland, not all stoats were removed from Secretary Island within the first year of trapping. Our results indicate that the stoat population is now being maintained at a very low number and, as a result of immigration and breeding by residual resident animals, there has been no further decline This latter finding may be related to the island's size. Many eradication programmes against mobile carnivorous predators have taken several years to reach completion. Examples include cats (*Felis catus*) (Bester *et al.* 2000; Veitch 2001; Algar *et al*. 2002), stoats (Crouchley 1994), and mink (*Neovision vison*) (MacDonald and Harrington 2003).

Recent home range estimates obtained for stoats on Resolution Island (home range diameter C. 486 m; Clayton *et al.* 2011) indicate that the decision to space trap lines at a distance no greater than 1400 m apart may have been an over-estimate of resident stoat home range. On Secretary Island, a few stoats may have retained very small home ranges, despite the significant population reduction, and have therefore failed to encounter a trap. Since female stoats have smaller home ranges than males they may be less likely to encounter a trap. Nonetheless, twice as many females as males were trapped. Alternatively, some stoats may avoid entering a trap tunnel either for extended periods of time, or even in perpetuity, as was the case on Maud Island, New Zealand (Crouchley 1994). On Secretary Island, stoat tracks were twice recorded in snow along ridgelines with traps present, which indicated trap avoidance. Based on Maud Island experiences, continued trapping can eventually eliminate stoats that have avoided traps for periods of up to several years.

Genetic data revealed enough variability across all loci to show some degree of differentiation between the mainland and original Secretary Island population, although F_{ST} values were relatively low. Differentiation between these groupings was supported by the STRUCTURE analysis which showed the data to be effectively split into two groupings. Evidence for immigration amongst the remaining stoats captured on Secretary Island after the initial year was from allelic differences (new alleles appearing) and from the assignment test using GENECLASS, which identified four first generation immigrants. There were also three individuals from that group which were assigned to the residential island population, while the remainder were unable to be assigned to either group, so were most likely admixtures from both.

The level of immigration detected from July 2005 to June 2008 was higher than predicted by Elliott *et al*. (2010) possibly due to beech masting in 2006 and a subsequent rodent and stoat plague on the mainland in Fiordland. During mast years, it is likely that there will be higher numbers of juvenile stoats dispersing from the mainland to inshore islands, such as Secretary Island. In February 2007, one stoat was caught on Seymour Island to the south of Secretary Island, the first in seven years of trapping. Another stoat was seen on Anchor Island, which had been free of stoats since 2001.

Further genetic work to include all of the stoats captured on the island since 2005 should help to refine the estimate for immigration. Molecular tools will also be used to determine the relatedness among individuals, thereby providing an estimate of population productivity; the absence of rodents on Secretary Island may mean that female litter size is reduced, which would explain why the number of stoats caught in summer on Secretary Island is not higher. King *et al.* (2003) demonstrated the significance of rodents driving population productivity in four beech forest sites in Fiordland. A shortage of rodents can lead to increased mortality of embryos and young in the den, while adult females remain healthy.

Low population productivity on Secretary Island strengthens the chances of eradication, which thus remains a key objective. A harsh winter, further refinements with the existing trapping programme or new technologies may hasten removal of the residual population. Moreover, stoat numbers have remained sufficiently low on Secretary Island to achieve anticipated conservation outcomes such as the reintroduction of several species of threatened birds (Wickes and Edge 2009). Monitoring species particularly vulnerable to stoats will be crucial in order to establish a stoat density threshold for future reintroductions, such as tieke/ South Island saddleback (*Philesturnus C. carunculatus*) proposed for 2015. The challenge is to detect stoats at extremely low densities without establishing a prohibitively expensive monitoring programme.

GENERAL CONCLUSIONS

Our programme was based on applying techniques developed on smaller islands over a much larger area. Although we planned to put all animals at risk of capture, this appears not to have been achieved, probably due to a broader range of habitat types than anticipated in the Secretary Island landscape. We also assumed that the level of reinvasion would be lower than preliminary genetic results have indicated. The experimental nature of this programme has opened the door for testing new ground in the field of island eradications and challenging some of the previously held views of what should and should not be attempted (see Edge *et al*. 2011). Molecular DNA tools have been invaluable in enabling managers to better understand what has happened on the island since the campaign began.

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The essential non-science of eradication programmes: creating conditions for success

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Abstract Among conservationists, eradication of non-native vertebrates is widely recognised as an often necessary step to restore island ecosystems and protect native biota. Less understood is the great difficulty of actually conducting an eradication programme. The biological and technical aspects of eradicating a population represent one category of challenge: how to ensure that every individual of the target species is removed, and how to know when that point has been reached. Here, however, we focus on a less appreciated but nonetheless essential category: how to put in place enabling conditions that will help ensure the success of the eradication effort. The planning and preparation required to conduct an eradication programme extend far beyond the realm of science and technical planning. Eradication programmes increasingly are multidisciplinary endeavours, requiring comprehensive financial, logistical, political, communications, and legal preparation. Without such dedicated support, sponsors and managers of eradication programmes introduce additional risk to an already risky investment of limited conservation resources, because even minor delay or interruption of the programme can have significant ramifications. Here, we provide an overview of the extent of planning and preparation undertaken to implement one of the most intensive efforts to date to eradicate an insular population of feral pigs.

Keywords: California, feral pig, island conservation, planning, risk management, Santa Cruz Island

INTRODUCTION

Non-native vertebrate species can devastate biological and cultural resources of islands and eradication is often necessary to remove the threats posed by these animals (Reaser *et al*. 2007). However, eradications can be logistically complex, expensive, and controversial, and can represent high risk investments of scarce conservation resources; multi-year, multi-million dollar investments can be jeopardised if even one individual escapes detection and enables the population to re-establish. For an eradication to succeed, it must meet predetermined conditions of success and have a solid scientific and technical foundation to its strategic and tactical approach (Parkes 1990; Morrison *et al*. 2007). Such projects must also have a solid foundation of operational, administrative, legal, communications and other types of support. These "non-scientific" aspects of an eradication project are important for very biologically-based reasons: once initiated, an eradication campaign must not be interrupted, lest progress in reducing the unwanted population be lost.

Managers of eradication efforts generally recognise that projects risk failure due to the difficulty of detecting animals at very low abundance. That risk can be reduced through strategic planning and implementation of the eradication project (Morrison *et al*. 2007). Deploying sustained pressure on the population in a systematic and intensive manner reduces the likelihood that animals will escape detection, reinvade areas already cleared, or replace those removed via reproduction. That in turn enhances the likelihood of ultimate success, and may reduce the overall cost of the project as well as the number of animals that ultimately need to be dispatched (Morrison 2007).

Given the importance of being systematic and intensive, it is crucial that eradication attempts, once begun, are sustained to completion. Even slight delay can compromise the programme. Interruptions can stem from a variety of factors: funding shortfalls, accidents, breakdowns in logistical support, legal intervention, and loss of political or public support. Interruption in an eradication project can enable replacement of the population through redistribution and reproduction, and so a loss of accomplishment to date. When the effort is reinitiated, it could require substantial reinvestment to return to previous levels of population reduction. Making up lost ground can be expensive, perhaps prohibitively so. If animals were able to reproduce because of the delay, the consequence will be even more animals ultimately needing to be eliminated. And those animals already eliminated would have died without any long-term conservation benefit.

Failed eradication attempts can incur substantial costs including not only the direct expenditures on the eradication effort (e.g., those paid to the eradication service provider), but also the indirect costs of administration and operations by the eradication sponsor and manager. Opportunity costs can also be high, because conservation funding and capacity invested in a failed eradication could have supported other restoration or biodiversity conservation initiatives. Failure may also have significant "reputational" consequences, and not just for those conducting the eradication but for the conservation tool itself, with effects that transcend the specific project. Failure of a high profile eradication effort could erode support for eradication programmes as a tool for conservation, making managers and funders less willing to invest in eradication efforts again or elsewhere. Failure could therefore have a cascading ecological cost: the biodiversity conservation outcomes needed on the subject island would not be attained, and the outcomes needed on other islands might not be attempted. Failed eradications can fate native species to extinction.

Thus, managers undertaking eradications must do so with an explicit focus on reducing the myriad risks of failure. Indeed, a principal responsibility of the sponsors and managers of an eradication project is to ensure that once launched it will be carried through to completion. As we outline below, that requires a focused, multidisciplinary support team – working well in advance of the actual on-the-ground effort – tasked with creating robust scientific, legal, administrative, and financial foundations for the project. As every eradication effort will encounter unique challenges and circumstances, it should be anticipated that projects will not go wholly as planned. The ability to implement adaptively requires a broad foundation of support.

Here we describe the support system developed for the eradication of feral pigs from Santa Cruz Island, approximately 40 km off the coast of Santa Barbara, California, USA. The Nature Conservancy (TNC) owns 76% of the 250 km^2 island and the United States National Park Service (NPS) owns the remainder. We do not describe

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the methods of the hunting and monitoring component of the eradication project (i.e. the eradication effort); those are described elsewhere (Parkes *et al*. 2010). Rather, we describe the role of the sponsors and managers of the project in creating and sustaining conditions that allowed the eradication effort to proceed unimpeded. We discuss the process by which the project was planned, and how it was supported. Although our example is an eradication project on an island, the principles would apply to pest eradication projects generally. This case study illustrates the extent of support demanded of an eradication effort of this scale, and as such may provide a model for reducing investment risk in future eradication efforts.

METHODS

To increase the chances of successfully eradicating feral pigs from Santa Cruz Island, we developed a foundation of internal and external resources that would provide support through the various phases of planning and implementation. In doing so, we sought input from others with past experience of similar projects that could be applied adaptively to our situation. We tried to anticipate circumstances that could arise that would impede implementation, and prepared accordingly. Below, we outline the general components of that foundation, illustrated with specific examples from Santa Cruz Island. We first describe key roles that needed to be performed in the planning and implementation. We then discuss where we focused our preparation to ensure that, once underway, the eradication effort would be resilient to disruption.

Clarifying roles

Eradication projects differ fundamentally from other management and restoration programs: if the targeted population is to be reduced to zero, a very intensive and specialised campaign must be sustained uninterrupted. Because eradication projects are complex and multidisciplinary undertakings, it is important to clarify the various roles and responsibilities of those involved so that accountabilities are clear. Basic functions were categorised as follows:

Sponsors: initiate the eradication project and ensure that the conditions for success are in place, e.g., funding, environmental compliance, contract management, communication with stakeholders.

Providers: conduct the on-the-ground eradication effort; in our case, a contractor with specialised expertise in the techniques we needed.

Managers: control resources and logistics, and serve as the on-the-ground support for providers.

Analysts: provide expert counsel in planning and monitoring, e.g., initial assessment of the feasibility of meeting an eradication goal and independent audit of progress during implementation.

While other important roles could be described (e.g., "external champions" that lend support for the project at critical moments, such as independent scientists, supporting organisations, and community leaders), our emphases here are the "core" functions. We do not suggest that each of these functions is exclusive. For example, on Santa Cruz Island, both TNC and NPS performed the roles of sponsor and manager. Similarly, the provider (Prohunt, Inc.) had a key role in planning and analysis, in addition to conducting the eradication. Generally, "providers" conduct the actual eradication field work, which for our project is more fully described by Parkes *et al*. (2010). An example of a role of the "analyst" in our project was evaluation of the probability that eradication had been achieved (see Ramsey *et al*. 2009). Below, we focus on the responsibilities we assumed as sponsors and managers of the eradication project.

Designing a "resilient" project

Expertise from many disciplines was needed to ensure that once initiated, the project would withstand disruptions and reach completion. The following were key elements of those foundations.

Scientific foundations

Scientific principles were not only important for the technical planning, implementation and monitoring of the project; they were also the basis of many of the non-science foundational components, such as our communications and legal strategies. Key components of the science foundations included:

Describing the threats posed by the target species: Well in advance of the actual eradication, we documented the extensive damage caused by pigs, based on published literature, observations, and inference (NPS 2002).

Understanding management options and preparing to defend the preferred method: We evaluated potential strategies that might achieve the desired conservation outcomes and were prepared to justify why we selected eradication by means of hunting over others (such as sustained control, translocation, and contraception.)

Developing an eradication plan: Once it was determined that pigs needed to be eradicated, we developed a plan that would address logistical challenges specific to Santa Cruz Island. External "analysts", e.g., from Landcare Research (New Zealand), played a key consultation role to ensure the planned approach was feasible and represented best practice.

Assessing and mitigating possible adverse effects of eradication effort: The motivation for undertaking an eradication is to protect resources, so it follows that there should be measures to minimise adverse non-target impacts during and after the project. In our project, examples of such precautions included: inspecting all areas where ground disturbance was planned (e.g., due to installation of a pig trap) for presence of sensitive plants or archaeological resources; using only non-lead ammunition; and reducing risks to the endangered island fox (*Urocyon littoralis santacruzae*) posed by the presence of hunting dogs (e.g., all dogs underwent a vaccination and quarantine regimen, and fox aversion training.)

Monitoring and managing the ecological response of eradication: Monitoring is crucial not only to detect and mitigate anticipated and unanticipated adverse effects (Morrison 2007), but also to maximise learning from the eradication project. Clear hypotheses and pre-eradication baseline data on key systems or taxa can leverage the research opportunity. Our monitoring also included biological samples from the pigs in case of future questions about whether certain wildlife diseases had a reservoir in the pig population. These data will also be useful if pigs reappear on the island and we need to ascertain whether they derive from the original island population or from a new release (e.g., resulting from sabotage).

Documenting effort of the eradication project: Recording all hunting and monitoring effort and outcomes (pig dispatches) using GPS units aided the day-to-day decision making of the provider, generated evidence of performance for the sponsors, facilitated coordination of activities by the island managers, and allowed for quantitative audit near the end of the project.

Contractual foundations

Contracting for eradication efforts poses unique challenges, in part because of the intensity and flexibility required in implementation and the degree to which it relies on coordination with the managers and analysts. Furthermore, it is impossible to know with certainty whether the provider has completed the eradication until sufficient time has passed without detection. Here the interests of the sponsor and provider may diverge: the sponsor might prefer withholding a substantial final payment to minimise risks to its overall investment, but doing so might not be financially realistic for provider. Meanwhile, the provider may prefer maximal payment up front to have the resources to mobilise an intensive initial effort. An important element of the contracting process was thus a fair and appropriate distribution of risks. This in turn required each party to understand and reconcile the needs and constraints of the other.

We sought to establish a fixed-price contract with a provider having demonstrated expertise and a long-term professional commitment to eradication projects and conservation outcomes. We considered the provider's experience and reputation to be crucial. When a provider begins to report that animals can no longer be detected, sponsors need to have confidence in the professional judgment of the provider's team and trust that the project was implemented in a manner that did not simply make remaining animals harder to detect (Morrison *et al*. 2007).

A fixed-price contract structure, versus one based on time and cost reimbursement, set in place incentives for efficiency that likely reduced the duration and cost of the programme (Morrison 2007). The provider's eradication plan for implementing the project was translated into a project timeline that could be incorporated into an enforceable contract. The contract outlined a framework for a general sequence of activities structured around performance milestones to which incremental payments would be pegged. Because eradication projects are idiosyncratic, even the most seasoned provider will face uncertainty as to how the actual eradication will transpire; time, effort, and cost are just estimates. All those involved understood that implementation would be necessarily adaptive within the contracted framework and that the contract would need to be amended periodically as the project progressed.

Legal foundations

Environmental compliance, permitting, and administrative process: The importance of strict and documented adherence to the regulatory compliance process is difficult to overstate, as the adequacy of environmental review can be a basis for legal challenge. The National Park Service was responsible for environmental analysis of project alternatives, impacts, and mitigations, in compliance with the National Environmental Protection Act (NEPA). This process included public review and resulted in the decision that eradication was the preferred alternative for protecting the natural and cultural resources on the island.

Legal preparation and defence: Individuals and/or organisations opposed to the goals or methods of the project may at any time mount a legal challenge. In addition to careful adherence to the compliance process, we proactively discussed all proposed work with legal counsel, so that defence teams were ready to engage if needed. Preparation included identifying experts in many disciplines willing to serve as resources should we need to quickly respond to a challenge.

Ethical foundations

Some people believe that killing sentient animals is unacceptable, even for preventing extinction of other species. Still more people are likely uncomfortable with the notion of killing large numbers of animals. To maintain support for eradication programmes, projects must be planned, conducted, and communicated in a way that demonstrates attention and sensitivity to these issues. The projects must also focus on reducing, to the extent practicable, the stress and suffering of target (and non-target) populations. A strong ethical foundation requires conducting due diligence on alternative methods, and being able to articulate how animal welfare has been incorporated into project activities. Hiring highly skilled marksmen to implement the project was a key component of our efforts to meet standards for euthanasia of wildlife (AVMA 2001).

Community foundations

Community support for an eradication has two components: support for the project during its implementation, and help with protecting the investment once completed (e.g., partnering to prevent reinvasion). In our project, the social dimensions of eradication may have been less complex than on sites where there are resident human communities. Even without a resident population on Santa Cruz Island, there were still community groups with direct or indirect interests in issues associated with the eradication. We therefore conducted public meetings to discuss the project, and focused direct outreach to Native American representatives with ancestral connections to the island and to user groups (e.g., boating clubs) with active ties to the island.

We also recognised sport hunters as a major constituency that we did not want to alienate against our pig management efforts (e.g., by advocating wildlife agencies to oppose the eradication). We therefore coordinated with the State of California to offer a rare public hunting opportunity on the portion of the island owned by TNC. This was conducted well before the eradication so that there would be no residual effects of the "recreational hunt" on pig behaviours that would compromise the "eradication hunt" (see Morrison *et al*. 2007).

Several animal protection organisations expressed concerns about the project, specifically questioning the need to eradicate pigs. TNC and NPS tried to maintain open communications with these groups. Although we did not expect them to become project supporters, we had the goal of showing that the project was based on a serious assessment of environmental impacts and the methods and contractors were chosen to minimise the suffering of individual animals.

Because eradications can have a high media profile, appear controversial, and often require direct and or indirect governmental support, political engagement in the relevant arenas of government was a priority. In order to respond to the needs of elected officials, we gave regular briefings on issues and progress.

Communications foundations

Strategic communications and outreach: Well before implementation, we developed outreach strategies to build the necessary internal and external support for the project. This involved identification of the individuals and entities important to inform about or otherwise involve in the effort, and effective delivery of information to them. In addition to individually tailored outreach to key partners, funders, and community leaders, our communications programme involved a proactive media strategy with information that was fact-based, constructive, and educational. We hosted opportunities for media to visit the island, discuss the project, and meet key staff. We also prepared media materials with frequently asked questions (FAQs) and other background information.

Two elements of our communication approach were especially important. First, we used messages that simplified the complexity of the eradication effort so that the project rationale was easily understood. Our primary emphasis was project outcomes: this was not just about killing pigs; it was about keeping the island fox and numerous rare plants from going extinct. Second, we were especially careful with the language we used to discuss the project. We focused on the science, and avoided terms that were emotionally charged or potentially insensitive. Because numerous entities were involved in the project, we invested considerable effort in developing and providing consistent messages. We provided guidance and training to key staff, including the pig hunting team, on how to effectively communicate and represent the project.

Internal communications: We developed crisis communication protocols that identified points of contact, internal communication channels, and delegations of authority. We also did not assume that all within our respective organisations were supportive of the eradication effort—or even aware of it. So, we conducted internal outreach to brief staff, answer questions, and outline instructions as to whom to direct inquiries regarding the project.

Information management: Information management during the eradication was essential, especially for safety. We were concerned that if details about the specific location of hunting activities found their way to opponents of the project, it might attract civil disobedience and so compromise the safety of the hunters as well as the protesters. We were therefore disciplined in our exchanges of information among the various personnel and partners involved in this project, making sure that documents, emails, photographs, maps, and so on would not be problematic if they found their way into the public arena.

Financial foundations

Because eradication projects can be expensive, providers must have the resources required to succeed. Funding for the whole project needs to be committed before the job is begun, and accessible as needed. In our case, project funds came from private (TNC) and public (NPS) sources.

Operational foundations

Dedicated institutional capacity through the planning and implementation: Planning and implementation of eradication efforts requires disciplines ranging from project administration to media and governmental relations. Orchestration of that effort required dedicated personnel, with the skills and capacity necessary to advance the project and address problems that arose. From the onset of the project, senior management of TNC and NPS made it clear to staff that there was no higher priority than success of the eradication and to organise and prepare accordingly.

Infrastructure, facilities, and equipment: The eradication team required considerable logistic support before and during the project. Prior to the eradication effort, for example, we needed to install over 43 km of high-tension pig exclusion fencing to divide the island into smaller management zones. Improvements or upgrades were also needed for on-site housing and roads, power, water, and communication systems. We needed reliable information management systems to allow efficient downloading, backup, and analysis of project data. Adequate housing and facilities had considerable bearing on the maintenance of morale of the hunters, which surely affected their performance in the field and the attainment of our overall goal.

Safety: Human safety was the paramount consideration in all aspects of this project, not just among the hunters but for all island users. While the eradication was underway, there was still the full array of island activities on the island including research, resource management, maintenance, and recreation. We therefore needed to manage access and coordinate activities so that users would not interface or interfere with the hunt, and vice versa.

RESULTS

While the on-the-ground phase of the eradication effort took place between 2005 and 2007, efforts to establish the enabling conditions for the project were underway for years prior. The environmental compliance process was initiated in 1999, and culminated with the completion of the environmental impact statement in 2002. The search for a provider for the eradication service was conducted via a competitive Request for Proposals issued in 2004; and in 2005 Prohunt, Inc. was selected.

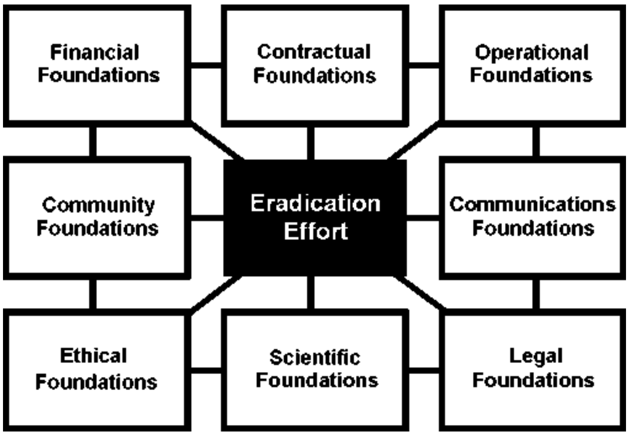

Fig. 1 A framework for resilience in eradication projects. When eradication sponsors and managers create adequate foundations of support for the project, they can buffer the eradication provider from disruptions that might compromise the on-the-ground effort.

The work described above created a support structure for the eradication project that enhanced its resilience to expected and unexpected challenges (Fig. 1). Below we highlight ways in which those foundations were tested. Some challenges were anticipated, others not. All required creativity and institutional agility to troubleshoot and resolve.

A capacity to adapt was required from onset of the project. Many of the provider's employees, for example, were not US citizens, and securing visas and firearm importation permits was unexpectedly protracted, which in turn forced modification of the mobilisation schedule. Following Hurricane Katrina in 2005, the federal budget to NPS was cut, which resulted in a 50% reduction in NPS boat service to the island. This affected our planned transport of personnel and equipment. Other surprises were biological: as the eradication effort mobilised, we discovered numerous eagle nests on the island; hunting efforts needed to be greatly curtailed in the vicinity of those nests until they were no longer active. Technological issues also surfaced. For example, we faced considerable challenges getting the radio- and GPS-collars for the pigs to perform reliably; much effort was spent in "R&D" and less in actual application – again precipitating a need to modify plans and amend the contract.

The implementation sequence also required flexibility. The eradication was designed and contracted to progress systematically from west to east across the island. But at the time of contracting, we could not know how long that progression would take. The easternmost zone was the portion of the island most accessed by Park visitors. As the programme advanced, we realised that unless the planned progression across the island was modified, active hunting would be underway during the peak visitor season. Disruption to Park visitors such as park closures could undermine community support for the project. We therefore modified our plan (and the contract) to advance that area of the island in the schedule, concentrate the provider's efforts in that zone, and thereby reduce the disruption to visitors. Fortunately, the contract structure, and the commitment of the providers to the needs of the sponsors, meant that such amendments were straightforward.

Before and during the eradication effort, editorials in the nearest mainland newspaper consistently opposed the project, even publishing names and photographs of key personnel involved (e.g., Santa Barbara News-Press 2006). Our outreach to media before the eradication, however, helped ensure that the full conservation story was communicated broadly and well ahead of controversial coverage that accompanied the eradication (e.g., MSNBC 2005).

Throughout the project, we faced legal challenges from animal rights interests petitioning to have the project stopped, mostly based on allegations of inadequate environmental compliance process (e.g., US District Court 2005). Fortunately we had invested significantly in legal preparation. For example, we were able to quickly assemble formal declarations from subject area experts to address each of the plaintiff's complaints. Our preparation was perhaps most tested when a former superintendent of Channel Islands National Park unexpectedly published an essay in a local newspaper suggesting that the NPS environmental review process was flawed (Setnicka 2005). Although his accusation was not supported by the formal administrative record (US District Court 2006), it did create issues that needed prompt attention so that public support and our legal position would not be compromised. All told, we faced five successive legal challenges, all of which were rejected by the court.

Our hunting dogs provided a final illustration of the need to expect the unexpected. We imported 23 trained dogs to the island. Each dog had to undergo an extensive vaccination and quarantine regimen due to concerns of introducing canine pathogens or parasites to the endangered island foxes. Protocols were developed by a team of wildlife veterinarians with years of experience in island fox conservation management issues. Midway through the eradication project, one dog dug from his kennel into that of another in oestrus, and soon thereafter she produced a litter of pups. This revealed a deficiency in our biosecurity protocols: some parasites of concern can remain in cysts in mammary tissue and be released upon nursing. Had the whole dog team become re-infested, it could have prevented their use in the field and significantly disrupted the project. Again the veterinary team was mobilised to develop revised treatment protocols for the dogs so that risks of transmission to foxes could be contained. We also made it impossible for one dog to dig to another's kennel! Had we not established a network of collaborators and advisers on the project and been able to mobilise a timely response, even something as seemingly benign as puppies could have compromised the programme.

DISCUSSION

The Santa Cruz Island feral pig eradication was completed in an unprecedentedly short time for an island of its size; the interval between the dispatch of the first and last pig was only 15 months (Morrison *et al.* 2007). While that is a clear testament to the skills and dedication of the hunting team, what enabled that accomplishment was the meticulous preparation preceding the actual implementation and the subsequent sustained comprehensive support by the sponsors and managers. This support ensured that there were relatively few surprises during implementation. It also helped us be prepared for and respond to the surprises that did arise.

Clarity about roles and responsibilities throughout the planning and implementation was essential. Simply put, a key role of TNC and NPS was to ensure that providers were able to focus on their job without disruption or delays. Delegations of responsibility among the multidisciplinary teams were clear, and communication was frequent and effective. Interestingly, once the provider was selected and the contract signed, the relationship between contractor and contractee quickly became a conservation partnership. A team ethic permeated all: we were committed to a common goal of eradication, and recognised that we were wholly reliant on the others excelling in their roles if we were to achieve it.

This case study highlights how it is not enough to plan an eradication based on biological and logistical considerations alone. Even though the scientific justification for removing feral pigs from Santa Cruz Island was compelling (NPS 2002), the preponderance of evidence that eradication was necessary did not beget eradication. Eradications are conducted within a social and political context, which may affect their feasibility to the same extent as biological factors. Our project required, in addition to technical planning, massive logistical coordination, public and private fundraising, garnering of political support, communications and outreach, and more. These "non-science" aspects of the eradication effort were an essential complement to its scientific underpinnings.

Every eradication project is unique and the strategies that we used to prepare this project may differ from those needed elsewhere. Because funding is limited, eradication teams need to assess the extent to which they invest in proactive versus reactive risk management strategies. Our emphasis on proactive strategies was influenced by Santa Cruz Island's location adjacent to millions of southern California residents, its status as a National Park, co-owned by a high profile international conservation organisation, and the level of opposition to previous eradication efforts on neighbouring islands (e.g., Los Angeles Times 2002).

In that context, we found extensive outreach to key stakeholders – including potential project opponents – to be essential. Projects in other contexts, like islands that are more remote or that have permanent residents, may assess risks, costs, and opportunity costs differently than we did. What we underscore is the importance of risk management decisions and contingencies that reflect the unique challenges confronted by each eradication project.

Lessons from this case study can be applied to reduce risks inherent in eradication efforts. In the face of a global biodiversity crisis and extreme global change, it is imperative to increase the pace and scale of eradication programmes against invasive species, particularly on islands, so that ecosystems can gain greater resilience to future stresses. The past decades have seen a marked increase in the sophistication and rigour of eradication projects (Veitch and Clout 2002; Veitch *et al*. 2011). Those experiences, combined with better understanding of the full complement of skills and functions necessary to conduct successful eradication, should help to scale up and accelerate restoration efforts and so the conservation of highly imperilled biota.

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Running the gauntlet: advocating rat and feral cat eradication on an inhabited island – Great Barrier Island, New Zealand

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Abstract Great Barrier Island is one of the largest inhabited offshore islands in the New Zealand Group; it is scenically spectacular with large areas of regenerating forest, and rare species of plants, reptiles and birds. Many of New Zealand's worst introduced mammalian pests are absent from the island, but mice (*Mus musculus*), rats, (*Rattus rattus and R. exulans*), rabbits (*Oryctolagus cuniculus*), pigs (*Sus scrofa*) and feral cats (*Felis catus)* are present. The island has C. 800 permanent inhabitants, but numbers are swelled in summer by 'off-island' house-owners and visitors, whose expenditure enhances the economy. Recognisable factions in the population have divergent views on the importance of biodiversity conservation and sustainability in the future economy of the island. Since 2003, the Great Barrier Island Charitable Trust (GBICT) has advocated the eradication of rats and cats from the island, with a strategy that aims to educate and involve the local community in related conservation and educational projects, while lifting the biodiversity profile of the island within administrative agencies. This paper outlines these methods and discusses some of the successes and set-backs encountered.

Keywords: Rodent eradication, Great Barrier Island Charitable Trust, Windy Hill Rosalie Bay Trust, community-led conservation

INTRODUCTION

Successes with eradications of invasive species from uninhabited islands have inevitably led to consideration of eradications from those that are inhabited. Regardless of potential benefits to biodiversity, unless these benefits are understood and supported by local communities, eradications may be actively opposed. In New Zealand, the complete eradication of all introduced mammals has been achieved on at least 80 uninhabited islands of up to 11, 000 ha, with identified benefits for numerous native species of plants and animals. Many of the remaining large islands have resident communities. On Great Barrier, feral goats (*Capra hircus*) have already been eradicated, but other pests remain. In this paper we examine the perceived attitudes of islanders to the removal of the worst remaining pest species and local issues that will need resolution if this is to proceed.

Great Barrier is a large offshore island in northern New Zealand (ca 27,400 ha). The terrain is mostly rugged , clad in 'scrub' or forest, with steep slopes and cliffs along most of the coast. The 'scrub' is dominated by canopies of manuka (*Leptospermum scoparium*) and/or kanuka (*Kunzea ericoides*), and represents areas cleared of forest during the early days of European exploitation. Much of the scrub is now in transition to native broadleaf or conifer dominated forest (Ogden 2001).

The island has escaped introductions of some of the most serious introduced mammalian pests of New Zealand including possums (*Trichosurus vulpecula*), mustelids (*Mustela erminea, M. furo, M. nivalis vulgaris*), red deer (*Cervus elaphus*) and Norway rats(*Rattus norvegicus*). However, three species of rodents (*Rattus rattus, R. exulans, Mus musculus*), rabbits (*Oryctolagus cuniculus*), pigs (*Sus scrofa*), and feral cats (*Felis catus)* are present. Extensive areas have been grazed in the past by feral goats and cattle (*Bos taurus),* but goats have been eradicated and cattle remain only in small areas. Several endemic endangered New Zealand birds and reptiles are present. Great Barrier remains a national stronghold for brown teal (*Anas aucklandica*), kaka (*Nestor meridionalis*), black petrel (*Procellaria parkinsonii*), and chevron skink (*Oligosoma homalonotum*). The lack of some serious pests and the presence of some significant rare species, provides the basic rationale for proposals aimed at elimination of rats and feral cats. The potential economic benefits to the community, now largely reliant on tourism with a strong outdoor recreation component, are also considerable.

The island currently has C. 800 permanent human inhabitants in several small communities served by ferries and aircraft from Auckland, the nearest large city. Numbers are swelled in summer by 'off-island' house-owners, and visitors. This population can be divided into four main groups, albeit with some overlaps: 1) holiday home owners who live and work elsewhere; 2) permanent inhabitants living on the island but without deep Island roots; 3) members of the early farming families, born on the island ('the settlers'); and 4) descendants of the original Maori inhabitants, mainly members of the Ngati Rehua hapu of Ngatiwai. These groups have differing perspectives on environmental issues, conservation, land-use, and island governance. Agreement with one sector may therefore generate a reverse effect from another.

In this paper we outline the activities of two Charitable Trusts, operating as non-governmental organisations (NGOs), that promote and practice pest control on Great Barrier Island. NGOs such as these have a role in complementing the activities of established governmental agencies. Collaborations can be achieved by reaching the wider community and fostering locally derived solutions to issues that are of direct interest (e.g., Berkes 2004). The Great Barrier Island Charitable Trust (GBICT), has advocated the eradication of rats – and other mammalian pests – on Great Barrier Island since 2003 employing the strategy of conservation and education initiatives outlined in this paper. The Trust has the following vision statement:

'To protect native species through the eradication of rats and feral cats, to re-introduce species lost to the Island, and to work towards building an ecology-based economic framework for Great Barrier Island'

The Windy Hill Rosalie Bay Catchment Trust (WHRBCT) was formed in 2001 with the aim of improving biodiversity by reducing rat, cat, and feral pig numbers to facilitate natural breeding of native birds and re-introduction of species lost to Great Barrier. The ecosystem benefits of rodent control at Windy Hill are described by Ogden and Gilbert (2009). The Windy Hill Project has provided a research arm for the GBICT.

This paper describes the activities of the GBICT, outlines outcomes achieved, and the nature of opposition to our goals. In presenting our case history, we emphasise the most successful approaches and lessons learned in the belief that this will be useful to others planning eradication campaigns on inhabited islands.

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ADMINSTRATIVE CONTEXT

Island infrastructure and supporting agencies

The island infrastructure is administered by Auckland Council. The interface between the islanders and the Council is provided by a locally elected Board. About 68% of the island is public reserve administered by the Department of Conservation (DOC), which has a base on the island. Policy oversight for activities conducted by DOC is provided by the Auckland Conservation Board. The island is within the Hauraki Gulf Marine Park, oversight of which is provided by the Hauraki Gulf Forum, comprising representatives of all relevant statutory agencies and Maori groups.

Funds for infrastructure, such as roads, wharves, garbage disposal, etc. are obtained from an annual levy (rates) by Auckland Council on all landowners, referred to hereafter as ratepayers. Other infrastructural facilities such as walking trails and protection of threatened species are provided by funds allocated by the government to DOC, with priorities set after consultation with community groups and the Conservation Board.

Charitable trusts

Charitable trusts are bodies set up for specific non-profit purposes under the legal requirements of the Charities Commission. The GBICT comprises seven trustees, the newsletter editor, and 120 members. Members receive an annual report and a quarterly newsletter (*GBI Environmental News*), which is also distributed free to all island residents and off-island rate-payers. The Trust facilitates information flow between the various conservation groups on the island (Fig. 1), and statutory agencies including DOC, and Auckland Council. The activities of the trust are supported by grants, subscriptions, and donations.

Fig. 1 Great Barrier Island showing locations mentioned in text, and the main community-based trusts engaged in pest control and/or habitat restoration.

The WHRBCT is based around the Windy Hill Rosalie Bay catchments at the southern end of the Island. This trust comprises four trustees, one of whom is the project manager. Since 2001, the WHRBCT has been engaged in a programme of weed, rat, feral cat, feral pig, and goat control, reintroduction of species (robin, *Petroica australis longipes*) and research. The area trapped/baited for rodents and feral cats now comprises 620ha with C. 5000 bait stations on 80 km of cut tracks. This Trust employs four full time and two part time employees funded primarily by grants.

MAIN ACTIVITIES OF THE TRUSTS

From the start, GBICT recognised the necessity of underpinning its vision with sound science, and of communicating with all segments of the island community. Two main types of activities, which are not mutually exclusive, have been undertaken by the Trust: the transfer of information to the community; and research.

Research activities

The 'referendum'

In 2006, The Trust organised an Island-wide questionnaire which became known as the 'referendum' (Fig. 2). This was intended to inform the trustees on the degree of support for/against the aims of cat and rat eradication. The questionnaire was sent to 1800 residents and ratepayers and replies were received from 585 (32%), of which over 300 were island residents; a proportional response by residents of approximately 40%.

The questionnaire asked for 'yes' or 'no' answers to the GBICT continuing to research, and work towards, the elimination of feral cats, and to research the ecological and economic benefits of a rat-free Great Barrier Island. An accompanying explanation gave the vision statement and stated that the questionnaire was not a proposal to go ahead with an eradication plan, and that there would be no further action unless it was supported and led by the Great Barrier Island community.

Over 90% of respondents supported continued research and "working towards" feral cat eradication, and 93% supported more research on the ecological and economic benefits of a rat-free Great Barrier.

Many of the returned questionnaires were annotated with comments and questions which were answered in subsequent issues of *GBI Environmental News*.

Bird counts

During 2006 and 2007, GBICT organised five-minute bird counts at 16 locations throughout the Island. The purpose of these was: 1) to provide information that might be of comparative use should rodents and feral cats be eliminated; 2) to teach local people about bird identification and ecology; and 3) to engage them in discussion about the Trust's aims. Other bird observation activities were linked to the counts, such as a survey of bittern (*Botaurus poiciloptilus*), spotless crake (*Porzana tabuensis*), and kingfisher (*Todiramphus sanctus*), and a count of beach-wrecked birds. These activities were supported by a grant from the Biodiversity Advice Fund administered by DOC. A separate series of counts were made of kaka (*Nestor meridionalis*), again involving local people. The second of these counts was planned to coincide with similar counts made on the mainland (*www.kakawatchnz.org*), thus linking Trust activities with wider interests.

Bird counting activities involved at least 78 members of the Great Barrier population on five occasions over two years. Results of the bird counts were summarised by Ogden

(2009), and outlined in issues of the *GBI Environmental News* distributed to all residents. This activity increased the Trust's profile in the community and was regarded as a positive activity by most people.

Information transfer

Tiritiri Matangi Island trips

In summer 2005-06, GBICT organised three one-day trips from Great Barrier to Tiritiri Matangi Island, from which Pacific rats (*Rattus exulans*) were eliminated by aerial application of brodifacoum by Supporters of Tiritiri Matangi and DOC in 1993. This island is 40km from Great Barrier and a showcase restored island, with strong volunteer input to tree planting, species translocations, and maintenance (Rimmer 2004). The aim of these visits was to invite selected 'opinion makers' in the Great Barrier community to see biodiversity conservation achievements in the absence of rats. Trip participants completed a questionnaire about the relevance of the trip to their understanding of the Trust's vision. These trips involved 48 members of the community, and questionnaires were answered by all 33 persons on the first two trips. They were not distributed on the last trip.

The questionnaire had some questions asked on the outward boat trip, and some on the return to assess what information was gained or changed during the day.

Only eight of the 33 respondents had visited Tiritiri Matangi before, which indicates that this conservation success story, although nearby, is not well known on Great Barrier Island. The worst pests on Great Barrier were ranked as rats> feral cats> rabbits. Some respondents did not consider pigs to be pests. Asked to indicate (on a five point scale), their response to the statement: "It is very important to make Great Barrier Island pest free", everyone marked either: "1. Strongly agree" or "2. Agree".

Knowledge of Great Barrier's endangered birds was poor, although their conservation was supported enthusiastically by almost everyone. The bird species best known were brown teal (*Anas aucklandica chlorosis*) and robin (*Petroica australis longipes*), clearly indicating the value of the publicity given to robin translocations to Windy Hill and Glenfern Sanctuary in 2004 and 2005.

Most of the respondents knew, or assumed, that poisons had been used to eliminate rats from Tiritiri Matangi, but only two people (6%) knew that an aerial drop was the method used. Natural history aspects of conservation (birds, vegetation) were consistently ranked more highly than socio-economic aspects. Comments indicated ambivalence to increased tourism on Great Barrier Island and a widespread view that the relationship between DOC and the public of Great Barrier needed improvement.

Three guidelines were gained from these trip questionnaires: 1) the role that birds could play in persuading people that pest eradication is important; 2) the general lack of knowledge about toxins and their role in New Zealand conservation; and 3) the need to address economic aspects of conservation, and specifically rat eradication.

The Environmental News and State of Environment Report 2010

GBICT has spent more time collating data about the ecology/economy of Great Barrier Island than on primary research. The data collation has enabled articles in the *GBI Environmental News,* letters to the local newspaper (*Barrier Bulletin*), and material filed in the local library. This work culminated in 2010 with the publication of a 200-page

REFERENDUM

Please answer the questions below and return in the pre-paid envelope by (date).

CIRCLE THE ANSWER YOU WISH TO AGREE WITH

1. Do you support the GBI Trust continuing to work towards the elimination of feral (wild) cats on GBI?

YES / NO

Please note that we are ***not*** *concerned here with domestic pets, although in the event that a plan to eliminate feral cats is initiated in future it would be necessary to have a system of registration and all domestic cats neutered.*

2. Do you support the GBI Trust continuing to explore the ecological and economic benefits of a rat-free GBI?

YES / NO

Please note that you are ***not*** *voting to support either rat or feral cat eradication at this point. We assume that you would want more information before doing so. You are voting to support our efforts to continue to research the pros and cons. When we have more information on the economic aspects and the actual feasibility of the eradication process we will present that to you and ask again!*

Fig. 2 The referendum document. An explanatory document accompanied this form; see text.

"*State of the Great Barrier Environment*" report. A 22-page abridged version was delivered free to all residents and ratepayers, and the full version made available on the internet (www/gbict.co.nz), in the local library, to selected agencies and to all Community Board members.

The quarterly newsletter, *GBI Enviromental News,* is a sixteen-page magazine covering topics relevant to conservation on Great Barrier. Accounts of the Trust's activities, and the results from projects such as the bird counts, are presented. The Newsletter is aimed at a general Great Barrier Island readership, and 1200 copies are printed and distributed to off- and on-island ratepayers. It has been our most important means of communication, and is well regarded by most recipients. It is distributed free of charge, using grant money.

Open days, public lectures and workshops

'Open days' at Windy Hill, Glenfern Sanctuary, Morton's farm property near Awana, and a day trip to Kaitoke Swamp (Fig. 1), were designed to inform the community about activities of various trusts, and/or to allow discussion of conservation issues between trustees and the public. These were attended by 20 to 60 people.

In 2006, GBICT initiated a series of public lectures on New Zealand conservation, especially endangered birds and pest control. The 'Summer Lecture Series' comprised lectures on the economic and social aspects of invasive species in the Pacific region and the effects of rats on endangered New Zealand birds. Other public presentations on the birds of Great Barrier, and wetlands have been delivered in conjunction with DOC.

Workshops on methods for rat control around properties were organised in conjunction with the Windy Hill Rosalie Bay Catchment Trust, at the three main settlements. Speakers from the Biosecurity section of Auckland Council participated in these events. Workshops were designed to generate local practical involvement in rodent control. Judy Gilbert presented the Windy Hill rat trapping, baiting and tracking tunnel results, and demonstrated practical aspects of rat control. These workshops stimulated one local rat control programme, which subsequently ceased, and a pest management initiative led by local Maori at Motairehe, which is still functioning. Support was also given to rat-trapping by children at Okiwi School, in the nearby forest reserve.

Liaison with other groups

The GBICT has had Community Board and DOC representatives at its meetings since 2008, and communicated its vision to local Maori, including a presentation at the Motairehe Marae. Trustees also participate in activities organised by DOC. The community-based rat-trapping programme at Tryphena was initiated with DOC support. The GBICT has also given support to other conservation projects, such as the pest eradication on Motu Kaikoura Island, a predator-proof fence and associated activities at Glenfern Sanctuary, and the Katherine Bay Restoration Trust on iwi land. Liaison with other groups has been an important component of the Trust's activity, culminating in a meeting of all interested parties in 2009. This meeting was organised and coordinated by the DOC, as a prelude to future networking meetings. The *State of the Environment Report* (2010) also constitutes a transfer of information between the Trust and other community groups.

DISCUSSION

Here we examine some responses to GBICT within the community, external influences on perceptions about pest control, and the role of communities in restoration initiatives.

Publicity positions of news media

From 2003 – 2005, GBICT was a regular contributor to the local newspaper, *Barrier Bulletin*, running a "Rat Chat" column, and publishing letters on topical aspects of conservation. The paper at that time provided a useful outlet for our vision of a pest-free island It was not until after publication of the (supportive) referendum results in 2006, that any negative comment arose. It was claimed that the Trust was planning World Heritage Status for the Island, that it advocated aerial applications of poisons, and that it would impose costly biosecurity and quarantine measures at wharves. These measures would impinge further on the rights of landowners.

This negative comment escalated from letters and newspaper editorials, to the banning of GBICT members from some land areas, and local body opposition to pest management suggestions. Attempts to clarify issues, or correct erroneous statements attracted further misinformed opposition.

As a result, the Trust decided to withdraw from further public debate through the media. We now present our views in *GBI Environmental News*, and use other news media only to advertise our public activities. This 'lower profile' approach may have been partly successful, most of the original antagonists probably still oppose our vision statement, but there is now some support on the Community Board. Not everyone is convinced of the damage done by rats and feral cats, nor of the potential economic benefits should these pests be eliminated, and the debate now centres around the potential use of toxins.

The important conclusion from the 2006 experience was that, despite enthusiasm and strong science backgrounds, the Trust entered into the political arena without adequate planning or awareness.

Information flow problems

Two processes have resulted in a faction of Great Barrier residents becoming strongly opposed to any suggestion of rat and cat eradication.

The first is that some people read our suggestion that rats and cats could possibly be eradicated as a fact that they would be eradicated. They then promoted that as fact and concluded that there would be a mass distribution of aerially applied toxin. This therefore bypassed our ability to discuss options and built a faction opposed to our suggestions. This faction also mostly opposed a perceived increased biosecurity and dismissed any suggestion of economic benefits from rat eradication.

A second factor was a film dealing with the aerial application of compound 1080. This film was professionally presented, but contained many errors of fact, statements taken out of context and a fundamental mis-understanding of experimental techniques applicable to ecosystem management. The film was clearly intended to generate support for the banning of the aerial application of 1080 in New Zealand, and was shown to island residents with the inference that this would happen on the island. This effectively undermined the otherwise improving DOC/public consultation process on Great Barrier, and provided ammunition for the anti- GBICT faction. The use of 1080 has in fact never been suggested for rat and cat eradication. We know of no avenue to counteract such deliberate distribution of misinformation, except to keep on stating the truth.

The "bottom-up" approach

Our approach to research and information transfer rested on the assumption that eradications cannot be carried out on inhabited islands without strong community support. On Great Barrier Island, the community has until recently been rooted in a resource exploitation ethic centred around farming, mining and logging (Armitage 2001). This early community probably had little awareness of the unique biodiversity of New Zealand, or the special role of pest-free islands in this respect, and could not have afforded some of the conservation measures we now take for granted.

Because it is an island, and inevitably somewhat isolated in consequence, these views appear to have been slower to change than elsewhere in New Zealand. However, with increased levels of communication (television, internet) and travel (especially tourists and holiday home owners who live off-island), views are changing, and a polarisation is evident. Currently there is no objective assessment of these views, which of course differ over different topics. The Trust's 'Referendum' and other related unpublished polls seem to imply strong support for investigating the feasibility of rat and feral cat eradication. On the other hand, letters and responses in the *Barrier Bulletin* indicate opposition. The Community Board has not yet agreed to support a feasibility study.

CONCLUSIONS

The Great Barrier Island Charitable Trust has considerably advanced ecological understanding and environmental awareness on Great Barrier Island. However, progress towards the main goal of rat and feral cat eradication has been slow. This is partly because of different attitudes to conservation in a segment of the Great Barrier community, and partly because of a failure, by the trustees, to perceive the importance of the power-structures on the Island.

It is also unfortunate that the editor of the main newspaper, the *Barrier Bulletin*, has opposed the Trust's vision. Our own publication *The Environmental News*, has gone a long way to counteract this opposition, and has been the most successful strategy we have employed for raising awareness of these issues in the community.

Participatory activities, such as bird counts and the trips to Tiritiri Matangi Island, have been more effective in communicating our vision than have passive activities, such as guest lectures. The latter cannot be very effective until there is an interested audience to attend them. Personal discussions between the GBICT trustees and members of the community are certainly the most effective way of explaining our vision, but they are time consuming and can be exhausting.

Further progress will involve gaining the support of the Community Board, and outside bodies, such as the Hauraki Gulf Forum and the Auckland Conservation Board. Our completed State of the Environment Report has been supported by these bodies, and may lead to more bottom-up support. Once a groundswell of support can be demonstrated, the statutory authorities appear ready to recommend a full-scale study of the feasibility of rat and feral cat eradication on Great Barrier Island.

ACKNOWLEDGEMENTS

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Estimating the duration and cost of weed eradication programmes

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Abstract Two prerequisites for realistically embarking upon an eradication programme are that cost-benefit analysis favours this strategy over other management options and that sufficient resources are available to carry the programme through to completion. These are not independent criteria, but it is our view that too little attention has been paid to estimating the investment required to complete weed eradication programmes. We deal with this problem by using a two-pronged approach: 1) developing a stochastic dynamic model that provides an estimation of programme duration; and 2) estimating the inputs required to delimit a weed incursion and to prevent weed reproduction over a sufficiently long period to allow extirpation of all infestations. The model is built upon relationships that capture the time-related detection of new infested areas, rates of progression of infestations from the active to the monitoring stage, rates of reversion of infestations from the monitoring to active stage, and the frequency distribution of time since last detection for all infestations. This approach is applied to the branched broomrape (*Orobanche ramosa*) eradication programme currently underway in South Australia. This programme commenced in 1999 and currently 7450 ha are known to be infested with the weed. To date none of the infestations have been eradicated. Given recent (2008) levels of investment and current eradication methods, model predictions are that it would take, on average, an additional 73 years to eradicate this weed at an average additional cost (NPV) of $AU67.9m. When the model was run for circumstances in 2003 and 2006, the average programme duration and total cost (NPV) were predicted to be 159 and 94 years, and $AU91.3m and $AU72.3m, respectively. The reduction in estimated programme length and cost may represent progress towards the eradication objective, although eradication of this species still remains a long term prospect.

Keywords: Branched broomrape, eradication feasibility, *Orobanche ramosa*, stochastic dynamic model

INTRODUCTION

One requirement for eradication is that sufficient funding is available to complete the programme (Myers *et al*. 2000; Panetta 2009; Simberloff 2009; Gardener *et al.* 2010). For weeds, programme duration may be in the order of decades (Mack and Lonsdale 2002) owing, among other reasons, to the persistence of seed banks. However, weed eradication programmes have often been initiated without realistic estimates of the resources required to achieve their objective. This is understandable up to a point, because during the early stages of an incursion there may be uncertainty about the extent of spread and critical biological attributes of the target species. However, we maintain that subsequent reviews have often been undertaken without sufficient consideration of likely duration of the programme and hence future requirements for resources.

In simple terms, a weed eradication comprises the search effort required to delimit an incursion plus the additional search and control effort required to prevent reproduction until extirpation is achieved over the entire infested area (Panetta 2009). The feasibility of eradication depends upon such disparate factors as: 1) the number, area and spatial distribution of infestations; 2) detectability of the weed, and 3) biological characteristics such as time to reproduction and seed persistence (Panetta and Timmins 2004). Cacho *et al.* (2006) demonstrated the crucial effects of weed detectability and search effort on the duration of a weed eradication programme. They also showed that for a given level of detectability and search effort, search speed, control effectiveness, germination rate and seed longevity had the greatest influence on eradication programme length. Later work provided preliminary estimates of the cost and duration of eradication programmes that could be used to prioritise weeds for control (Cacho *et al.* 2007).

The attempted eradication of some major weeds in Australia has involved cost-sharing arrangements whereby the federal government provides 50% of total funding and the states and territories provide the remainder on the basis of the relative risk posed to each by the incursion (Panetta 2009). Major reviews of these programmes are undertaken at three year intervals, but tend to have an operational focus, without due regard to how long it might take to achieve the eradication objective and hence funding requirements over the long term. In this paper we present an estimate of the duration and future cost for an eradication programme against branched broomrape (*Orobanche ramosa* L.) in South Australia. We also demonstrate retrospectively how, on the basis of available information, estimates of both programme duration and cost can change over time.

METHODS

The eradication programme

Branched broomrape is an annual obligate parasite that has a wide range of broadleaved crops as hosts (Jupp *et al.* 2002). It has been estimated that in 2006 the annual value of Australian crops at risk from branched broomrape was approximately $AU1.87b (Econsearch 2008). An economic evaluation of an eradication scenario for branched broomrape suggested a benefit:cost ratio of 3.4 over 30 years. This assessment assumed that it would take 60 years for 100% infestation of susceptible crops and 15 years for a maximum yield loss (35% for all host crops) in any given area of infestation (Econsearch 2008). However, contamination of products with branched broomrape seed could have a major impact on export markets, since many of Australia's trading partners are free of this species. This was not factored into the analysis.

Branched broomrape was first detected in Glenelg, South Australia in 1911, as a single infestation that disappeared within a few years of detection. The species was not observed again until 1992, in the vicinity of Bowhill, 90 km E of Glenelg (Jupp *et al.* 2002) and was considered to have resulted from a separate introduction. This second infestation was eradicated by fumigation with methyl bromide, but over the next seven years, an additional 22 infestations were found within a 15 km radius. Broadscale surveys were then undertaken and in November 1999 a quarantine area covering all known infestations was declared in order to contain and eradicate the weed. A cost-sharing arrangement between the federal and state governments for an eradication programme was initiated in 2000 (Wilson and Bowran 2002).

Surveys between late winter and early summer have continued at yearly intervals within and adjacent to the quarantine area, as well as on properties in other areas with links to infested properties. The highest densities of branched broomrape's weed hosts inhabit the perimeter of paddocks, so searches target this area, with a few additional transects across each paddock (Jupp *et al.* 2002). Only about 3% of a paddock is searched each year (N. Secomb pers. comm.), which accounts for the low search cost when expressed on a per hectare basis (Table 1). The total area over which the weed is distributed is currently 7450 ha.

Table 1 Economic and associated information employed in model run for 2008.

Search ($AU/ha)	2.77
Area searched (ha)	333,000
Control ($AU/ha)	341.27
Area treated (ha)	1634
Administration ($AU)	532,831
Research and communication ($AU)	352,269
Discount rate	0.06

Infestations are controlled by a combination of host denial (including control of the weeds that are hosts for branched broomrape) and soil fumigation of roadside and smaller satellite infestations (Wilson and Bowran 2002). Although there is still some uncertainty regarding potential seed persistence for this species, the operational criterion for eradication of an infestation of branched broomrape is the lack of detection for 12 consecutive years (Panetta and Lawes 2005).

Records were acquired for each infestation for each year of the eradication programme from 1999 to 2008. In cultivated situations, infestations were defined by the total area of a paddock in which branched broomrape plants had been detected; in other situations they were defined by minimum convex polygons (IUCN 1994) that incorporated the outermost plants. Infestations were designated as active in any year that branched broomrape was detected. The total area of newly detected infestations was calculated for each year as was the total cumulative infested area. Records were also maintained of the area searched for each year.

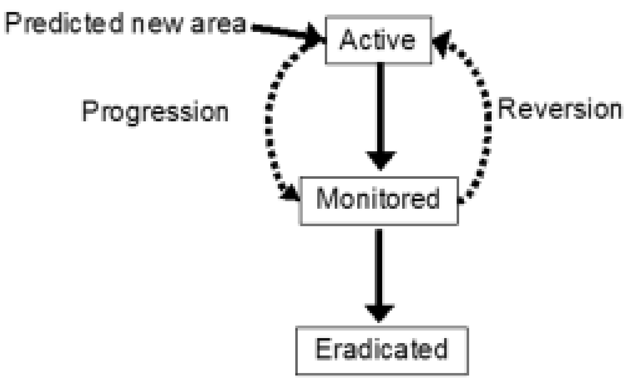

Fig. 1 Schematic diagram illustrating the functions upon which the stochastic dynamic branched broomrape eradication model was based. See text for description of the progression and reversion functions.

The model

Model structure

We developed a stochastic dynamic model (Fig. 1) for predicting the trajectory of total infested area, and hence programme duration. In this model, total infested area is divided into: 1) an active state in which the weed is detectable above ground; and 2) a monitored state where no recruits have been detected for at least 12 months (Panetta 2007). Data from the programme are used to estimate *progression* from the active state to the monitored state and *reversion* from the monitored state to the active state upon the further detection of plants. Given these transition rates, at the end of each time step the amount of infested area that is in the active or the monitored state is updated. When the weed has not been detected in an infestation for 12 years, the infestation is considered to be eradicated and hence the area of the infestation is subtracted from the total infested area. To date, however, there has not been sufficient time within the programme to eradicate any infestations.

The model is based upon three functions (Fig. 1):

1) The predicted discovery of new infested area

2) The rate of progression of infested area (considering all infestations) from active status to monitored status

3) The rate of reversion of infested area (considering all infestations) from monitored to active status.

Table 2 Categorisation of infested area relative to the time since last detection of branched broomrape for the three years for which the model was run. Note that zero years since last detection denotes active infestations and that the criterion for eradication is 12 years since last detection.

Years since last detection	Area (ha) 2003	2006	2008
0	4113	3150	1634
1	167	1134	1769
2	1097	345	871
3	886	831	1003
4	70.8	11.3	20.1
5	-	929	744
6	-	579	5.3
7	-	68.6	558
8	-	-	816
9	-	-	29.4
10	-	-	-
Total	6334	7048	7450

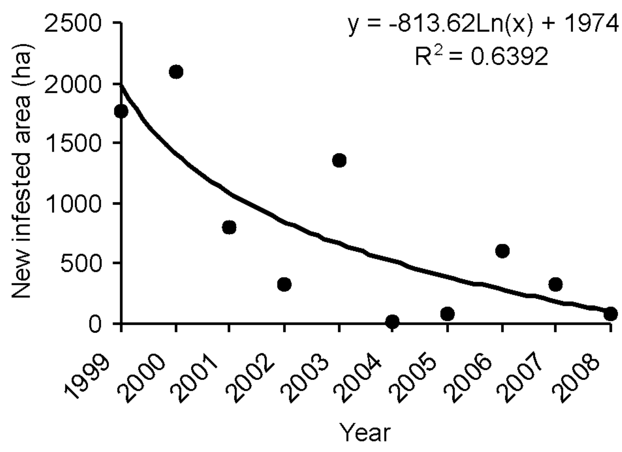

Fig. 2 Detection of new infested area during the course of the branched broomrape eradication programme.

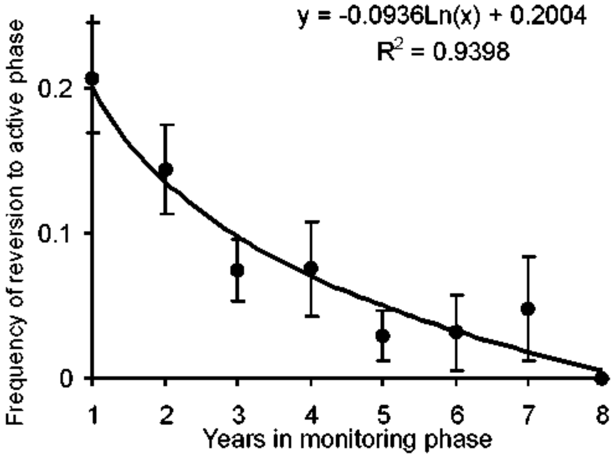

Fig. 3 Reversion from the monitoring to the active phase as a function of time in the monitoring phase for branched broomrape infestations. Bars represent standard errors.

Predictions of future detection of new infested area were based upon regression of historical data for detection of new infestations (Fig. 2).

The rate of progression from the active phase to the monitoring phase (0.696 $\pm$ 0.138, mean $\pm$ SD) was calculated from the data for all years (1999-2008) of the eradication programme.

Reversion from monitored to active status could be calculated only from 2001 onward, since the first year in which infestations could reach monitoring status was 2000. Thereafter, for each year and each stage of the monitoring phase (e.g., 1, 2, 3…*n* years since last detection) (see Table 2) the rate of reversion to the active phase was calculated by expressing the number of infestations reverting as a proportion of the total number of infestations in that stage. These rates were then regressed against the number of years without detection and the resulting relationship was used to model reversion of infestations from the monitoring to the active phase (Fig. 3).

The model simulates the active infested area at any time, calculated as:

$$A_t = A_{t-1} + A_n + A_r - A_p$$

where A_t = total active area at time t

A_{t-1} = active area at the previous time step

A_n = new infested area detected since the previous time step

A_r = area that has reverted from the monitoring stage to the active stage since the previous time step

A_p = area that has progressed from the active stage to the monitoring stage since the previous time step. Note that the area of any infestation that remains in the monitoring stage for a time step automatically advances to the next category of years since last detection (Table 2).

The model operates on annual time steps, corresponding to annual searches for the weed. It allows the user to specify both the maximum time period and the number of Monte Carlo simulations to be employed. Stochasticity was introduced by sampling randomly from a normal distribution based on the rate predicted by a regression equation. More specifically, the rates of change for a given iteration of the model were calculated for the three functions as: $y = \alpha - \beta\ln(x) + \varepsilon$, where (depending on the function) y represented new infested area, progression rate or reversion rate; x represented calendar year or years in the monitoring phase; and ε is an error term which is normally distributed with mean 0 and standard deviation σ [$\sim N(0,\sigma)$]. The values of (α, β and σ) estimated from the data were: (1974, 813.62, 475.25) for new infested area, (0.696, 0, 0.138) for progression rate, and (0.2004, 0.0936, 0.018) for reversion rate.

The model simulates the process for any given set of parameters given by the user rather than optimising an objective function. We specified a maximum time frame for simulations of 200 years with 50 simulations for the results presented herein. In order to determine how predictions might have changed through time, the model was run initially for 2008 and then for conditions existing in 2006 and 2003. Insufficient data were available to estimate functions 1-3 (above) prior to 2003, and 2006 represented a year in which new detections led to almost a 10% increase in total infested area (Panetta and Lawes 2007).

Economic data

Data on programme expenditure between July 2001 and June 2008 were used to calculate model inputs since complete data for the 2008/2009 financial year were not available. Given that we used average values (see below) over a relatively long period, this data deficiency was not expected to have a major effect upon the results. Expenditure was divided between the following activities: treatment, searching, administration, and research and communications. Average values of these allocations (Table 1) were utilised for the purpose of prediction of future programme costs and we assumed that relative allocation between the activities would not change through time. As of June 2009, total programme expenditure was $AU32,548,000 (P. Warren pers. comm.).

In order to make the results modelled for 2006 and 2003 comparable to those for 2008, appropriate deflation factors were incorporated to adjust all costs to net present value (NPV).

Table 3 Predicted costs (present value) over the duration of the branched broomrape eradication programme (from 2008 until completion) and breakdown of those costs in relation to programme activities.

	$AUm	%
Total costs	67.9	100
Control	36.3	53.5
Search	16.1	23.7
Administration	9.37	13.8
Research and communication	6.11	9.00

RESULTS

Given recent (2008) levels of investment and current eradication methods, the model predicts that on average an additional 73 years will be required to eradicate branched broomrape in South Australia (Fig. 4 A) at an average additional cost (NPV) of $AU67.9m (Table 3). Eradication was achieved in less than 100 years in all 50 simulations (Fig. 4 B). Estimates of programme costs varied between $AU63m and $AU75m (Fig. 4 C).

When the model was run for the circumstances in 2003 and 2006, the average programme duration and total cost (NPV) were predicted to be 159 and 94 years, and $AU91.3m and $AU72.3m, respectively (results not presented). These results suggest a significant improvement in eradication prospects from 2006 onward, which is likely due to decreases in the amount of infested area in the active phase (Table 2). However, it is clear that eradication of this species has been, and remains, a long term prospect.

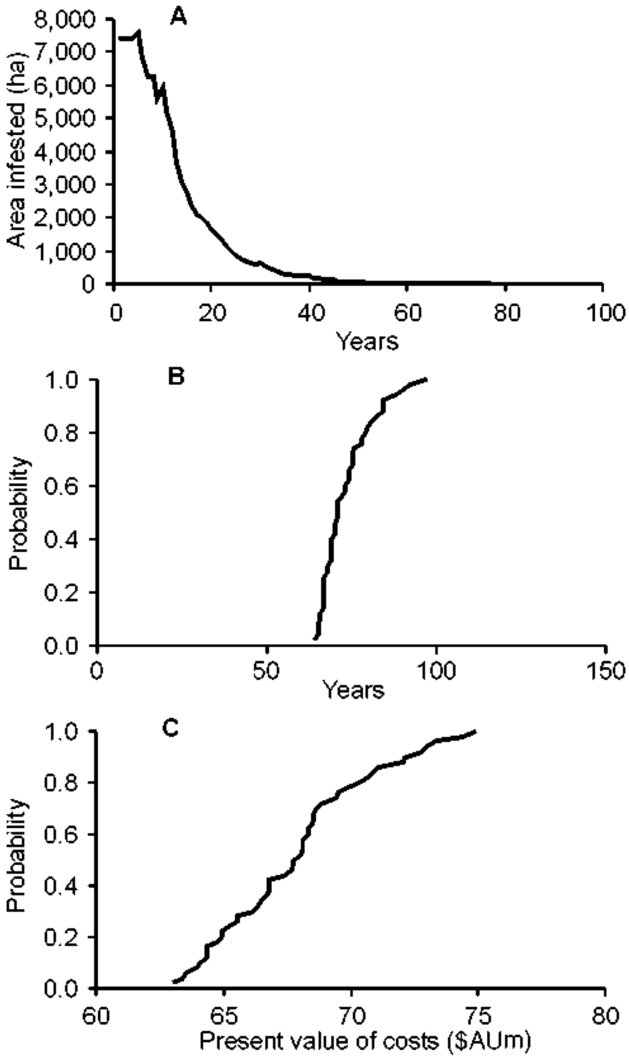

Fig. 4 Predicted trend in total infested area (sample run from 50 simulations) A and cumulative distribution functions for B time to eradication and C. total programme cost of the branched broomrape eradication programme.

DISCUSSION

Our estimates of programme duration and cost are probably conservative because we are not anticipating substantial increases in total infested area on the basis of current temporal trends (Fig. 2). Significant increases in newly detected area, and hence the pool of infestation in the active phase (Fig. 1), would extend the programme and incur substantial additional cost. In addition, the model is non-spatial; if infested areas are distributed through the landscape, programme costs would likely increase, particularly with respect to travel time.

It is worth considering the extent to which programme duration and cost could be reduced through improved management practices. If there are no more infestations, the rate of progression from active to monitoring status and the reverse transition (Fig. 1) become crucial components of the model. While relatively small areas can be controlled by fumigation, the most widely applied method of controlling infestations (and hence influencing their activity status) is host denial, which involves preventing the establishment and growth of the species that are parasitised. Cereal crops are not hosts to branched broomrape. Those broadleaved weeds that are hosts become effectively controlled while cereals are grown. However, it is difficult to control branched broomrape hosts without also eliminating the legume component in the pasture phase of cropping rotations. This is when it is most difficult to achieve progression to the monitoring phase and when reversions from the monitoring to active phase are most frequent (Panetta and Lawes 2007).

Eradication could be achieved more rapidly by directly targeting soil seed banks of this species, an approach used with success against another parasitic weed, witchweed (*Striga asiatica* L. (Kuntze)). By the end of 2007, witchweed infestations in the United States were reduced from 200,000 ha in the early 1970s (Eplee 2001) to approximately 900 ha (R. Iverson pers. comm.). As for branched broomrape, soil fumigants effectively killed witchweed seeds, but were too expensive for general use. However, when ethylene was used as a germination stimulant, and combined with treatments that prevented reproduction of the target species, it was possible to eradicate infestations of witchweed in about three years (Eplee 1992). A cost-effective method for rapidly reducing soil seed populations of branched broomrape would thus enhance the speed of eradication; this has been an area of considerable research activity in South Australia (Matthews *et al.* 2006; Virtue *et al.* 2006; Williams *et al.* 2006). Until such a method becomes available, however, the programme will remain largely reliant upon natural attrition of the seed bank, in combination with sustained prevention of its replenishment.

Even though our model predicts (on average) that 73 years would be required to achieve eradication, for the last 20 or so years, less than 10 ha of infested area may remain (see long tail of the trace in Fig. 4 A). There may thus be scope to shorten programme duration considerably through the application of expensive methods such as fumigation. This would lead to obvious savings across the various components of programme expenditure.

The allocation of future expenditure between different programme activities is based on several assumptions. For example, administration and the combined costs of research and communication have been treated as fixed costs. We also assume that high investment in control and searching is maintained throughout the programme. Some assumptions are perhaps easier to justify than others. It is unlikely that administrative costs would decrease substantially until at least the final years of the programme. While the need for research might decrease, there could be a compensatory requirement for increased communication so that public awareness and support are maintained through to completion of the programme. The cost of control is a direct function of the remaining infested area, so does not present much scope *a priori* for manipulation.

Whether searches over hundreds of thousands of hectares for new infestations will be required when only a few hundred hectares (or less) remain infested is debatable. To date there has been limited research on how to optimise investments in the search and control functions (e.g., Hester *et al.* 2008). Mehta *et al.* (2007) note that decision-makers often allocate fixed resources to certain activities over multiple time periods; these authors identify possibilities for updating management strategies through varying search effort over time. We believe that there is considerable scope for improving estimates of future costs of eradication programmes by exploring the potential effects of different temporal patterns of investment on both programme duration and cost.

Given the uncertainties that exist when a weed eradication programme commences, methods are needed to evaluate performance in conjunction with tools that can assist decisions to shift to alternative management strategies should these be warranted. Such decisions require quantitative measures that are utilised at predetermined decision points (Panetta 2009). Some measures of progress towards eradication have been developed (see Panetta 2007; Panetta and Lawes 2005, 2007). The present work adds to these by estimating costs associated with changes in the size and duration of the programme over time.

Feasibility of eradication must be considered in relation to the amount of investment (effort) available (Rainbolt and Coblenz 1997; Panetta and Timmins 2004; Panetta 2009). Increases over time in total known infested area will require increased funding, which has obvious implications for the ongoing assessment of eradication feasibility. The required investment should be estimated iteratively as a programme proceeds, and judgments made regarding whether eradication is still a feasible option given technical limitations and economic constraints (Panetta 2009). If properly informed, decision makers should be able to adopt a dynamic approach that allows switching to more economically optimal strategies (e.g., containment or sustained control) when required.

This study has quantified only the costs of branched broomrape eradication. A full analysis, which considered a 30 year period from the inception of the programme, estimated total incremental costs (NPV) of $AU75.46m and total incremental benefits of $AU258.52m. This yields a benefit:cost ratio (BCR) of 3.43 (Econsearch 2008). Interestingly, the BCR of a containment programme over the same timeframe was 3.85. Our model suggests that a BCR for the programme needs to be estimated over a longer timeframe but this is another exercise. The fact that an alternative management strategy is favoured economically in the shorter term suggests that eradication is not likely to be selected over longer periods, unless it remains advantageous to pursue eradication when potential negative impacts upon international trade are taken into account.

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What is required to eradicate red foxes (*Vulpes vulpes*) from Tasmania?

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Abstract The red fox is a major threat to middle-sized native vertebrates in Australia, many of which still thrive on the island state of Tasmania. Increasing evidence of the arrival of foxes in Tasmania since 1998 led the government to begin a campaign to intercept the invasion. An eradication programme began in earnest in 2002 but has not yet achieved its goal. The work of the Fox Eradication Program (FEP), as it was designated in 2006, was reviewed in 2009 and we summarise the main achievements and problems identified in that review. The feasibility of eradication was always uncertain because the island is large (6.3 million ha), foxes are cryptic and at very low densities, finding them is difficult, control methods are few (1080 poison baiting), and the methods do not provide direct evidence of success with a dead fox in hand.. Planning and practice had to be adaptive as the techniques for monitoring and control developed. The FEP has developed detection methods with estimated detection probabilities and now needs to integrate these systems with the deployment of the control to put all foxes at risk. Killing foxes that may survive baiting, and those potentially living in urban areas (where poisoning is difficult) or remote forest (which is assumed not to harbour foxes) remain as issues to be resolved.

Keywords: Detection, search, surveillance, validating eradication

INTRODUCTION

The red fox (*Vulpes vulpes*) was introduced into Australia in the 1860s and has spread over the mainland apart from the tropical north (Saunders *et al.* 1995). Foxes have caused extinctions of native animals between 35 g and 5500 g, and are the primary agent of decline for at least 77 vertebrate species listed as threatened on the mainland (DEWHA 2008). Neither dingoes (*Canis familiaris*) nor foxes reached the island state of Tasmania (Fig. 1), where there are still surviving suites of native species lost from the mainland. However, in 1998 several people saw a fox leaving a container ship at Birnie in the northwest of Tasmania, and in 1999 there were reports that foxes had been deliberately released in Tasmania (Saunders *et al.* 2006). These reports were followed by public sightings of foxes, which raised the possibility of their detrimental effects on 78 species of native Tasmanian vertebrates, including 12 already listed as threatened. In response, the Tasmanian Government formed a task force to attempt to eradicate the foxes. Their work began in 2002, was reviewed in 2003 (Kinnear 2003), in 2006 (Saunders *et al.* 2006), and again in 2009 (Parkes and Anderson 2009). This eradication is not a simple task. Despite a significant allocation of resources from State and Federal governments, evidence of foxes in Tasmania was continuing to appear in late 2009. In this paper we review why the task is difficult, and analyse with the advantage of hindsight what needs to be done to either improve the chances of successful eradication or, should that the task not be feasible, to set some change or stop rules.

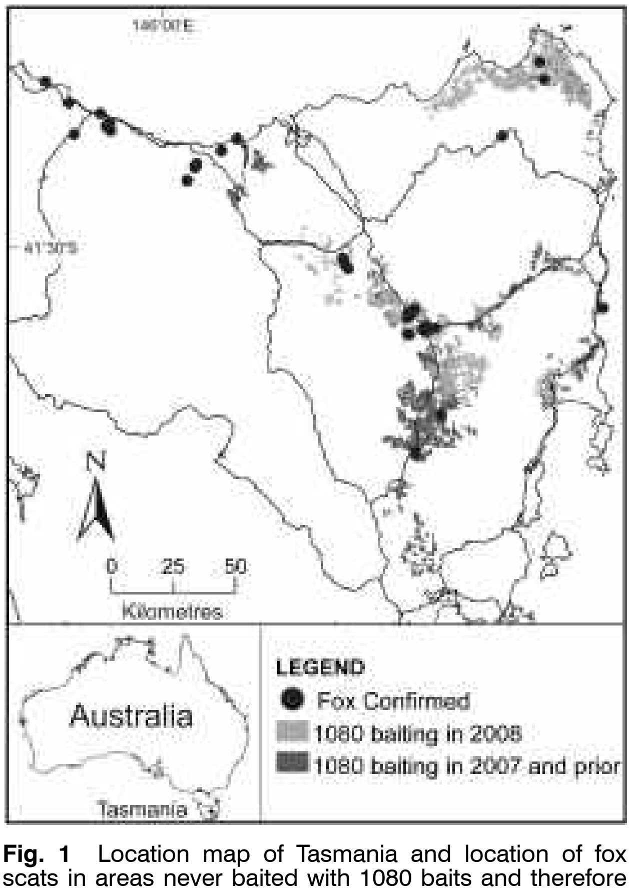

Fig. 1 Location map of Tasmania and location of fox scats in areas never baited with 1080 baits and therefore potentially never at risk as of late 2009. We assume a scat more than 1 km from a baited site meant that fox was not at risk, but without information on fox home range sizes in Tasmania this may be a pessimistic assumption.

RESULTS

The general problem

The Fox Eradication Program (FEP) faces daunting problems that means early assessments of the feasibility of eradication inevitably left large unresolved residual uncertainties and risks of failure:

Tasmania is large at 6.3 million ha. Half of the island is rugged, forested and remote, and the other half is rural and urban with a human population of 0.5 million.

Foxes are rare, cryptic, and hard to find. Some reports of foxes are unreliable (the public can mistake other animals for a fox, especially when glimpsed at night). Other detection methods are not instantaneous with lags between the certain presence of a fox and instigation of control at that site.

The behaviour and ecology of foxes in such colonising populations are unknown. Home range, dispersal, rates of increase, and potential Allee effects (the fragility of very low density populations due to chance events) that might lead to extinction of the population are all unknown and mostly unknowable for foxes in Tasmania.

Some Tasmanians doubted that foxes were present, despite the evidence from three foxes killed on the road and a fourth one that was shot. It was not until the development of faecal DNA tests in 2003 (Berry *et al.* 2007) that any rational doubt was allayed. Nevertheless, the dilemma of 'absence of proof versus proof of absence' argument

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remains a valid problem for the need to delimit the range of foxes and to validate the efficacy of any control. This issue typifies the end of all eradication operations (Ramsey *et al.* 2009, 2011).

Tasmanian managers of the FEP have relied largely on expertise and tactics available from mainland Australian states, where circumstances are quite different. On the mainland there are many foxes and fewer native prey, which makes extrapolation to a situation with few foxes and abundant prey a risky one. Although foxes are widely controlled in mainland Australia, the mind-set and practices of managers attempting sustained control may not always be appropriate for eradication. For example, the target for sustained control is to reduce the impact of the pest to some tolerable level, whereas the target for the eradication is to get the last one. In addition, there are few relevant precedents of fox eradication that could guide the Tasmanians. While Parkes and Anderson (2009) list 50 successful attempts, most do not resemble the Tasmanian problem.

At present, Tasmanian managers have only one effective control tool: baiting with compound 1080 (sodium monofluoroacetate). Elsewhere, trapping was also effective for the eradication of red and Arctic foxes (*Alopex lagopus*) in the Aleutians (Ebbert and Byrd 2002). Poisoning is not socially popular in Tasmania (Coleman *et al.* 2006), partly because it is perceived to place native non-target animals at risk. At a practical level, poisoning does not provide direct evidence of success if animals do not die on the spot. Baiting cannot usually be used immediately after a fox is reported because landowners have to be notified through a formal process and agree to allow the application of baits. In addition, 1080 baiting cannot be applied in urban or peri-urban areas.

All these uncertainties require managers to adapt their plans as they go along, which is not always simple when priorities change quickly but management structures are more difficult to modify in response. Uncertainty also creates unease among those funding the programme especially if success is not quickly achieved.

Locating foxes and delimiting their range

There have been more than 2000 public reports of foxes in Tasmania since 2002 (Fearn 2009, unpubl. FEP report; Parkes and Anderson 2009). An unknown proportion of these are in error as the public also reports seeing extinct thylacines (*Thylacinus cynocephalus*) and many reports of foxes (such as carcasses on roads) turned out to be other species when checked. The FEP grades the credibility of reports and checks those that are most credible or are from places of interest. For example, of the 32 public reports received in May 2009, only four were ranked as “excellent” by the FEP investigators.

In 2008, the FEP also deployed three dogs trained to find fox faecal scats, and tested the ability of these dogs and of people alone at finding scats (Parkes and Anderson 2009; D. Ramsey, unpubl. data). The dogs had between 10 – 40% chance of finding a fox scat known by the experimenter to be present somewhere in the 100-ha search areas and within a 30 minute search time. Teams of people searching for 300 minutes found a scat between 30 – 60% of the time. For a comparative search effort of 30 minutes, people found a scat less than 10% of the time. Operationally, such searches are made in response to a reliable public report, or as more planned surveys of areas of interest (‘hot spots’). Faecal scats also give false positives as the scats of other predators such as cats (*Felis catus*), dogs, and Tasmanian devils (*Sarcophilus harrisii*) can be visually mistaken for those of foxes. The dogs’ reactions do give some indication of reliability, but all scats are also tested for the presence of fox DNA. This test can identify individual foxes if the scat is fresh enough (Berry *et al*. 2007).

A stratified survey for fox scats began in 2008 to cover the half of Tasmania thought to provide the most suitable habitat for foxes during their establishment and colonisation. In all, 900 cells each 3 × 3 km were to be searched by people (without dogs) over three years (FEP, unpubl. data). In 2007/08, of more than 3000 scats found, seven (at four sites) contained fox DNA (Parkes and Anderson 2009).

As of January 2010, of 45 scats confirmed to contain fox DNA only 15 have been attributed to individual foxes (FEP, unpubl. data). No fox has been detected more than once from its scat. Assuming no error in the DNA testing, this finding creates some major uncertainties in the control campaign. First, the detection abilities of the dogs and people may be much lower than revealed in the trials. Second, the half-life of scats in the environment may be very short in Tasmania. Some scats may have been eaten by Tasmanian devils or buried by ants or dung beetles. Third, colonising foxes may be nomadic or have unusually large home ranges resulting in very low scat densities. If so, present searches are conducted at the wrong scale. All these issues are testable and the answers would inform managers on the optimal scales of both monitoring and control.

The current detection system has developed from mixed motives: to delimit the range of foxes or to locate individuals in order to deploy control and to prove foxes are present in the State to counter sceptics. The FEP’s efforts have sometimes been diverted away from the biologically essential delimitation and reactive control motives towards the politically necessary ‘proof of presence’ questions.

Deploying control

Foxes in Australia are usually controlled with dried meat or manufactured meat-based baits containing 1080. In Western Australia, these are aerially-sown because native animals in this State are not susceptible to 1080 (Twigg and King 1991). In the rest of Australia native animals are susceptible to 1080 so baits are buried to limit non-target risks (Saunders *et al.* 1996). In Tasmania, two main types of bait have been used: dried kangaroo meat baits and Foxoff baits. Both are buried to a depth of C. 10 cm and laid C. 200 m apart. Baits are flagged, logged by GPS and uneaten baits removed after 14 days to limit any risks to native animals and domestic dogs. Trials to estimate non-target risks showed this method to be acceptable because even if all baits eaten were taken by non-target native species and these animals died, the annual kill of about one death per 120 ha would not have any population effect.

In July 2002, a reactive strategy was implemented, which involved baiting all areas three or four times within a year after foxes were reliably reported. It is unknown how effective single or multiple applications of toxic baits are against foxes in Tasmania. However, on the mainland, about 10 days pre-feeding with non-toxic baits followed by toxic buried baits for about the same time can kill between 70% and 97% of foxes (Saunders and McLeod 2007). We assumed that the efficacy of the baiting on Tasmania would be less than on the mainland, given the abundance of natural food and the lack of pre-feeding with non-toxic baits.

Since 2006/07, about 1.2 million hectares have been baited with nearly 78,000 baits (Table 1). The decline in

Table 1 Baiting with buried 1080 baits for foxes in Tasmania since 2007.

Year ending April	No. baits buried	Mean % baits taken	Area baited (ha)
2007	10,953	NA	118,676
2008	40,156	18.2 ± 7.3	448,110
2009	26,724	10.9 ± 2.4	616,973

bait-take was significant but interpretation as evidence of fewer foxes (or fewer non-target animals) is confounded by changes in bait type.

The original reactive baiting strategy was only partially followed as areas were baited between once and seven times over several years. The reactive strategy as originally conceived had some problems. First, there were lags of up to 603 days between the reliable report of a fox or a scat and the application of bait (Table 2). This lag was caused by the time required to obtain landowner compliance and access to the area, and (for the scats) time required in the laboratory to validate the presence of fox DNA. Such lags were sufficient for foxes to move far away from the targeted baiting area. Second, there was a planning disconnection between the monitoring and control parts of the FEP. This may explain part of the above lag, but there were also 25 of positive locations of a fox where there was no reaction with control (Table 2).

The data also show that 61% of scats that were found lacked any control response (Fig. 1), partly because many of the scats were in the urban and peri-urban areas in the northwest of the island where baiting was not possible. It also revealed a planning issue to be resolved. If the FEP is to follow a reactive strategy, they need to react in space (bait where foxes are located) and in time to increase the chance that the fox is still present. An alternative approach to the reactive strategy and its attendant lags is to deploy baits under a precautionary strategy. The current baiting regime can cover up to 10% of the island in a year (Table 1) so it would be possible to deploy baits on rolling front(s) in some rational way (based on prior data on the presence of foxes or habitat risk analysis) across the island. However, this still leaves the problem of how to detect and deal with potential survivors of the initial baiting – and that is where a detection and reaction model can assist in planning a response and in setting success and stop rules.

Detection model for fox eradication

Like many eradication programmes, the FEP is data-rich but analysis-poor. The review proposed that the FEP use the data to inform management decisions on where to search for foxes (usually scats) and when to stop and declare success at a regional or island scale. These surveillance and stop rules can be done by quantifying the probabilities of a sequence of events that must occur to confirm the presence of a fox. First, a fox must in fact be present, it must defecate in the search area, the scat must survive, the scat must then be found, and its identity as a fox scat must be confirmed ideally via the presence of DNA. Bayesian analyses can be used to quantify these probabilities and used to inform the search efforts to achieve desired levels of certainty that no foxes found equals eradication and thus stopping rules for managers and funders.

The success of eradication can then be assessed in a small grid, at local scales, in areas where 1080 has been deployed, or over the whole island to give a probability that at least one fox persists given none are detected. Of course, to be 100% sure that no foxes are present one would have to look everywhere in Tasmania with a perfect detection system. However, as with all eradications, this is not possible. So managers have to set a probability at which they are comfortable – and that requires some analysis of the costs (in money, political embarrassment, damage to biodiversity) of falsely declaring success and stopping the programme too early. However, an additional advantage is that it does allow for risk analysis leading to some rational end point of the programme, which is something funders like; they rightly get nervous about open-ended campaigns that purport to be eradication.

Using existing estimates of scat-detection probabilities we can make some preliminary (and probably optimistic) predictions on the search effort necessary to achieve an acceptably low probability of fox persistence (i.e. successful eradication) if no fox scats are detected. For example, FEP managers could have a probability-of-persistence goal of ≤ 0.05, as was set in the Santa Cruz pig eradication (Ramsey *et al.* 2009), set some scenarios about the search effort based on fox habitat quality in each search cell (of say 1 km^2), and use the probability data currently available. A search of 20% of the cells in the highest risk areas without finding a scat would then meet the desired stop rule. We stress this prediction is based on a sensitivity analysis used by Parkes and Anderson (2009) to compare the relative probabilities of not finding a fox given one was actually present under different search scenarios. Obtaining a 'real' prediction would require better data on the parameter estimates in the model.

DISCUSSION

The FEP developed its strategies based on the best knowledge available from mainland fox ecology and control but was still faced with daunting uncertainties. To their credit, FEP managers have attempted to resolve these issues through a learn-by-doing approach and research focussed on: 1) improving safety to non-target animals, 2) the use of detector dogs and people, and 3) DNA analyses to validate fox presence. Learn-by-doing is more risky than formal adaptive management (Parkes *et al.* 2006)

Table 2 Baiting histories at sites where 41 fox scats have been located as an indication of whether the foxes are potential survivors of baiting, were potentially killed, or were never at risk.

Risk category	Number of scats	Time between baiting and scat location		Time between scat located and next baiting	
		Range (days)	Mean (days)	Range (days)	Mean (days)
Scat found in area previously baited (since 2006), i.e. a potential survivor	8	161–350	210 ± 52		
Scat found in an area subsequently baited, i.e., potentially at risk.	15 (includes 7 of the above)			0 – 603	142 ± 94
Scat found in area never baited since 2006, i.e., never at risk	25				

as the knowledge it provides can be unreliable. These characteristics may put the whole eradication campaign at risk if it takes too long and if the funders become nervous about the probability of success or find higher priority areas in which to invest. Aware of these risks, the FEP managers commissioned the 2009 review to describe what had been achieved, what had not been done and what must be done. The review noted the substantial amount of spatially-explicit data on fox locations and where control had been deployed. This meant that the use of Bayesian techniques could be used in this, and other similar projects, where zero pests is achieved by successive culls, to inform the uncertainty around when to stop.

The 2009 review showed how the monitoring component of the programme had drifted apart from the control component of the programme, as a consequence of following a reactive approach but with lags in the reaction time. If these time lags between detection and response cannot be resolved, moving to a precautionary baiting strategy, at least for the initial response, would allow the two components to be re-integrated.

However, under either the reactive or precautionary strategy, a major need for the programme is to develop a reliable alternative method to kill foxes that: a) may survive baiting for whatever reason and b) live in urban or peri-urban areas. Trapping, spotlight shooting, snaring and hunting have been tried without any success. The review suggested using trained predator detector dogs – not those trained to sniff out scats – to locate foxes in their daytime lairs so that they can then be killed by other means. So far as we know detector dogs have not been used to detect foxes but they are regularly used in eradication campaigns against other predators such as feral cats in Mexico (e.g., Wood *et al.* 2002), and stoats (*Mustela erminea*) in New Zealand (Theobald and Coad 2002).

A second residual uncertainty is that about half of Tasmania is dense temperate rainforest. On mainland Australia, such areas are not the usual habitat of foxes. However, these areas have no human population to report foxes and have not been surveyed in the scat detection systems. The probability that no foxes found equals successful eradication is thus lowered if there are gaps in the surveillance – by how much depends on the likelihood that the assumption is not true.

The review showed some areas where the FEP has made progress but also identified clear problems that have to be resolved. Funding agencies will take on risky projects as long as potential benefits are identified and the risks are clear. It is the lack of transparency that scares decision-makers when all can see that the task is difficult. A Parliamentary committee of the Tasmanian Government has just reviewed the FEP (Anon 2009) and despite the ongoing difficulties has accepted that the costs of failure are too high and has recommended that the programme should continue.

There is an additional lesson from this example. Planning paradigms for eradications that require successive actions or culls to reach zero numbers are intrinsically different from those such as aerial baiting for rodent eradication where there is a single intense period of activity. In the latter, the need is for meticulous plans that focus on getting everything right on the day (Cromarty *et al*. 2002). In the former, flexibility and change are required as events unfold and the best laid plans go astray (Parkes *et al.* 2010). Here probabilistic models are useful as part of managing these uncertainties and risks.

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Enhancing biosecurity at the Phoenix Islands Protected Area (PIPA), Kiribati

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Abstract The Phoenix Islands Protected Area (PIPA) of the Republic of Kiribati was established in 2006 and extended to cover 408,250 km^2 in 2008. The draft PIPA Management Plan aims to eradicate invasive alien biota (mainly vertebrates) from the top priority islands first then work towards eradication of invasive vertebrates from all eight atolls. Implementing improved biosecurity is crucial across the entire. Key risks identified via workshops and targeted consultation include potential invasions after visits by legal and illegal fishing vessels, tourist vessels and national freighters, any of which can carry a variety of invasive species. Key biosecurity approaches being implemented include passing a National Biosecurity Act, setting up a biosecurity committee, strengthened internal biosecurity as well as at the borders, and emergency response plans. A novel border approach involves the licensed international fishing vessels that visit Kiribati waters, where existing Government of Kiribati on-board observers can be trained in biosecurity and vessels fitted with geo-fencing radio-beacons. We propose that these vessels are required to be pest-free as part of licensing agreements. Surveillance and apprehension of other vessels will be through the complying captains reporting illegal vessels, together with the periodic deployment of aerial and sea surveillance craft. National freighters and other vessels will be inspected at ports of departure where biosecurity is also being strengthened, and also prior to entry at Kanton, PIPA. There is a need for further capacity development as well as international agreements with relevant countries at their departure ports. Our recommended biosecurity approaches are largely untested for Kiribati but will be continually refined.

Keywords: Invasives, surveillance, rats, cats, pigs, rabbits, McKean Island, Rawaki Island

INTRODUCTION

The Phoenix Islands of Kiribati in the central Pacific Ocean are isolated from other island groups in Kiribati by c.1000 km of ocean. The Phoenix Islands Protected Area (PIPA) was gazetted in 2006 and extended in 2008 to 408 250 km^2, which at that time was the world's largest marine protected area. The eight atolls have received little human settlement, and only Kanton is now inhabited. The plant communities on most of the atolls are little modified. The breeding seabird populations are globally important and comprise petrels and shearwaters (five species), storm-petrel (one species), tropicbirds (two species), boobies, (three species), frigatebirds (two species), noddies (three species) and terns (three species). The resident fauna includes two species of threatened seabirds: the Phoenix petrel (*Pterodroma alba*) and the white-throated storm-petrel (*Nesofregetta fuliginosa*), which are currently IUCN-listed as Endangered and Vulnerable respectively. The islands also provide important habitat for migrant species such as the bristle-thighed curlew (*Numenius tahitiensis*) (Vulnerable), Pacific golden plover (*Pluvialis fulva*) and other shorebirds. Islands in the PIPA are also important breeding grounds for green turtles (*Chelonia mydas*) (Endangered) and support many species of lizards and invertebrates, including the coconut crab (*Birgus latro*), and other species of land crabs.

The biota of the PIPA has, however, been depleted by the impacts of invasive species, particularly mammals comprising ship rats (*Rattus rattus*) Asian rats (*R tanezumi*) and Pacific rats (*R. exulans*), cats (*Felis catus*), pigs (*Sus scrofa*) and European rabbits (*Oryctolagus cuniculus*). A Critical Ecosystem Partnership Fund survey in 2006 indicated that seven of the eight atolls had been invaded by rats; only Rawaki has remained rat-free, enabling populations of Phoenix petrels, storm-petrels, shearwaters, blue noddies and others to maintain a foothold. However, Rawaki has supported rabbits for over 100 years where they have had serious impacts on vegetation and competed with petrels, shearwaters, storm-petrels and blue noddies for what little nesting cover remained. Meanwhile, large rats have arrived on at least two islands in recent years: Asian rats via a shipwreck on McKean in about 2001 and ship rats by unknown means and at an unknown date at Kanton (Pierce *et al.* 2006, 2010).

In this paper, we review the effects of mammal eradications in the Phoenix Islands to date, outline the biosecurity issues that threaten these and other proposed activities, and indicate how these issues are being resolved.

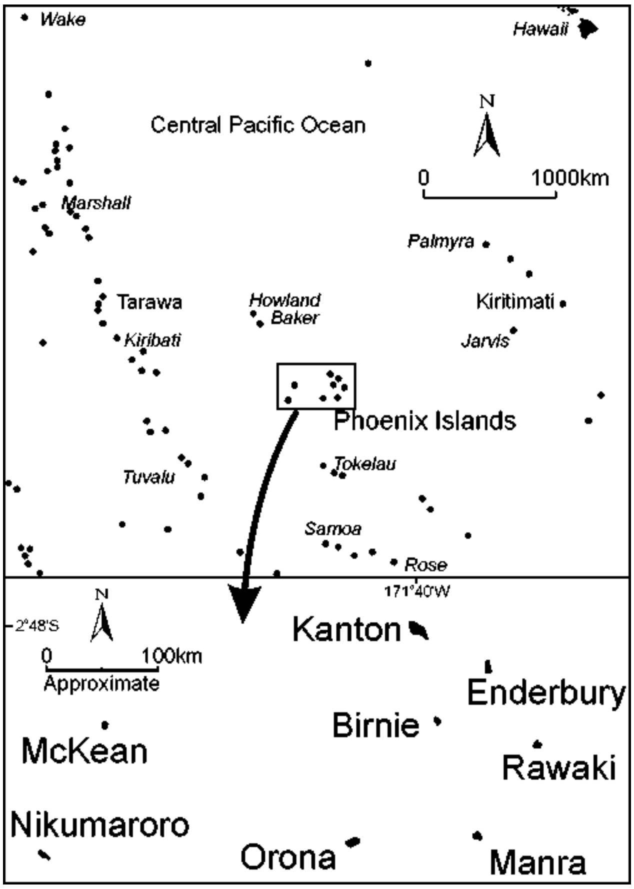

Fig. 1 The Phoenix Islands Group.

Table 1 Pest mammal status in the PIPA 2009.

Island	Approx. area (ha)	Pest status 2009	Comments
Rawaki	50	Nil	Rabbits eradicated 2008
McKean	30	Nil	Asian rats eradicated 2008
Birnie	50	Pacific rat	Operational planning underway
Enderbury	600	Pacific rat	Operational planning underway
Kanton	1100	Cat, Pacific rat, ship rat	Operational planning underway; inhabited island, major biosecurity issues, phoenix petrel etc colonies;
Orona	600	Cat, Pacific rat	Crab issues
Nikumaroro	500	Pacific rat	Crab issues
Manra	400	Cat, rat sp?	Crab issues, needs survey - pigs reported as well

PIPA RESTORATION TO DATE

The PIPA Management Plan (Government of Kiribati 2010a) identified atoll restoration via pest removal and biosecurity as a key objective. A first step towards this objective was achieved in 2008, when European rabbits and Asian rats were eradicated from Rawaki and McKean Islands respectively as part of a project funded and supported by NZAID and NZ Department of Conservation (NZDOC). Positive responses to these successful eradications were apparent 18 months later through changes in vegetation diversity and extent, and seabird productivity at both islands. For example, on Rawaki the shrubs kaura (*Sida fallax*) and *Portulaca*, which are now free of grazing pressure, are regenerating across the island despite a prolonged dry period. These shrubs provide greatly increased nest site availability and cover for frigatebirds, blue noddies, storm-petrels, petrels and shearwaters. On McKean Island, the nesting success of seabirds has increased significantly, notably amongst grey-backed terns and brown noddies, which had previously been losing virtually 100% of their eggs or chicks (Pierce *et al*. 2010). As well as providing local benefits for the PIPA, the recovering seabird populations will enable several species to potentially colonise other restored island groups in the central Pacific, either via natural dispersal or through artificial translocations.

Planning is currently underway to eradicate pests and restore additional islands, including Enderbury, Kanton and Birnie (Table 1). In addition, there is a crucial need to step up biosecurity measures at the PIPA and beyond to sustain the success of island restoration work. Seven of the islands are uninhabited and there are significant biosecurity issues that could lead to invasive species accessing the islands.

BIOSECURITY ISSUES FOR PIPA

Biosecurity issues in PIPA are similar to those elsewhere in the Pacific, but there are also significant differences and unusual risks. Particular risks are posed by uninhabited islands that are seldom visited by official parties, but which are in the vicinity of considerable risky boating traffic. Foreign specialists and staff of the Ministry of Environment, Lands and Agricultural Development (MELAD) have identified sources and mechanisms of

Table 2 Pest risk analyses and actions needed at pre-border and at-border sites.

Pathway	Source	Main risks	Prevention measures & other actions needed*	Responsibility
Very High Risk				
Illegal landings from people on Kiribati cargo boats that pass through the PIPA, and potential ship-wrecks of the same vessels	Tarawa, Kiritimati, and other northern Line Islands Cargo vessels are MV Matangare, Moomi, Mataburo, Betiraoi, Moamoa	Rats (several spp), mice, cats, dog, birds, ants, lizards	Government observer to be present on these boats to ensure non-landing compliance	PIPA/MELAD
			Provide bait stations, rodenticide and rat traps for permanent use by all captains*	Agriculture
			Inspect boats pre departure and on arrival at each of Betio (Tarawa), Kanton and Kiritimati and provide certification or quarantine as appropriate*	Agriculture
			Reinstate Quarantine/Biosecurity Committee to coordinate above measures and implement new regulations plus risk analysis under new Biosecurity Act. Improve boat hygiene to prevent accidental introduction of pests and monitor permitted/prohibited goods. Improve cargo regulations (prohibited/permitted product lists), cover packing materials and standards for fresh produce (e.g., fruit and vegetables). Regulations for male cats and dogs and restricted to inhabited islands of Lines and Phoenix.	MELAD/PIPA
			Port surveillance and control - currently focused on agricultural pests. Needs improving and broadening to cover rats, ants, cats. *	Agriculture
			Need inter-island regulations to be included under planned Biosecurity/Quarantine Act.	Agriculture MELAD
			Decide who is responsible for drawing up regulations.	PIPA
			No landing signage	PIPA
			Remove Enderbury coconut trees	PIPA

Table 2 continued

High Risk Pathway	**Source**	**Main risks**	**Prevention measures & other actions needed***	**Responsibility**
Legal fish boats (illegal landings, wrecks)	US mainland Korea, Taiwan Japan EU (Spain) Ecuador (Spain boats) NZ, China Am Samoa, Betio & Kiritimati offloading	Rats, mice, cats, ants, birds, reptiles (snakes)	International agreements for boat hygiene - none exists? Inspection at home ports by home country quarantine services? Inspection by Kiribati/observers - aim is 100% of vessels* Kiribati regulations - need developing to cover pests on board, powers of inspectors. Education & awareness in fisheries. Probably needs doing in home countries.	International Agencies International Agencies Fisheries & PIPA Fisheries Act. MELAD (& PIPA). MELAD, Agencies
	Pacific Is transit ports	Snakes? Unknown	Identify ports used. Then above measures apply.	Fisheries & PIPA
Illegal fish boats (illegal landings, wrecks)	IUU and others	Rats, mice, cats, ants	Observers on legal boats report these. Patrol boat and aircraft (Aust/NZ Orion). Get additional boat based in Kanton.	Fisheries GoK Maritime Command PIPA, CEPF.
Passenger/cargo & other planes (e.g., medical, surveillance) to Kanton	Australia, Hawaii, Kiritimati, Nadi, Tahiti	Rats, mice, snakes, lizards, mosquitoes, ants and other insects, frogs, toads, weeds	Form Tech Committee for Risk analysis. Include specific pests, permitted/prohibited product lists, packing standards, standards for fresh produce (e.g., fruit and vegetables), domestic animals, on-board treatments (e.g., residual insecticides). Draft pre-border agreements (different for each source country?) and seek pre-border agreement approval. Draw up regulations for airlines under planned Biosecurity Act. Implement regulations. Design improved quarantine procedures (including surveillance at airports for selected range of pests) and incorporate into regulations under planned Quarantine Act. Establish/improve quarantine (procedures including surveillance, facilities, officers) at Kanton & Kiritimati airports (and other airports in Kiribati).	Agriculture, SPC, SPREP, PIPA; ECD; outside input to risk analysis Agriculture (Quarantine), SPC, SPREP, PIPA; ECD. Agriculture; input from ECD, PIPA, SPREP, SPC. Ag - Quarantine Input needed from PIPA Committee, ECD, SPREP, SPC. Ag (Quarantine)

Moderate Risk Pathway	**Source**	**Main risks**	**Prevention measures & other actions needed***	**Responsibility**
PIPA Patrol boat	Tarawa, Kiritimati, Penrhyn	Rats, mice, ants	Maintain rodent bait station, inspect boat on departure (Tarawa, Kiritimati) and arrival (Kanton)*	Agriculture
Yachts (legal & illegal landings, wrecks) - < 50 applications per year.	Tahiti, Marquesas, Cooks Hawaii Kiritimati	rats, mice, birds, dogs, cats, lizards, ants, weeds	Review and possibly improve permit conditions. Improve inspection (procedures and training) in entry ports. Implement inspections in ports of entry (Kiritimati, Tarawa, Kanton, Fanning)	PIPA, ECD, SPREP. Ag (Quarantine). Ag (Quarantine).
Live-aboard tour boats (legal landings, wrecks)	Cooks Fiji	rats, mice, ants, geckos, insects, weeds	Update permit guidelines* Implement guidelines on permit. Inspections - observers on boats*	EcoOceania, SPREP, SPC. Currently rely on Captains. PIPA, Fisheries
Research & management boats (Naia, etc) (legal landings, wrecks)	Hawaii Samoa - Rarotonga	Rodents, snakes, lizards, mosquitoes, other insects, frogs, ants, weeds	Provide permit guidelines Update permit guidelines* Implement guidelines on permit. Inspections – observers on boats*	PIPA Technical input required as above. Currently rely on Captains. PIPA, Fisheries.

Abbreviations – Ag Agriculture division, ECD Environment and Conservation Division, MELAD Ministry of Environment, Lands and Agriculture Development, PIPA Phoenix Islands Protected Area, EU European Union, IUU Illegal, Unregulated and Unreported fishing vessels, CEPF Critical Ecosystem Partnership Fund, SPC Secretariat for the Pacific Community, SPREP Secretariat for the Pacific Regional Environment Programme,
* indicates details of recommended work being prepared in the Guidelines document.

potential pest invasions in the PIPA. This risk assessment was undertaken through workshops and meetings at Tarawa and included members of the PIPA management Committee, South Pacific Regional Environment Programme (SPREP), Secretariat for the Pacific Community (SPC) and ourselves, followed by subsequent discussion with key contacts.

Highest risks include, but are not limited to, rodents, cats, ants, other invertebrates, and seeds, being present on vessels and potentially invading PIPA islands via the following pathways: 1) passengers making illegal landings from domestic cargo ships; 2) personnel making illegal landings from fishing vessels; 3) researchers, managers and tourist operators making legal landings from vessels; 4) cargo off-loaded at Kanton or taken aboard at Kanton; 5) shipwrecks/groundings of yachts, fishing boats, cargo ships; and 6) air cargo arriving at Kanton in the future.

Recent steps to improve biosecurity include initiatives internally and at borders:

1) Kiribati Biosecurity Act, imminent (Government of Kiribati 2010b); 2) PIPA biosecurity guidelines being developed via CEPF funding; 3) Kiribati domestic freighters fitted with rodent bait stations and captains provided with bait; 4) monitoring by Agriculture staff at embarkation and destination ports; 5) a PIPA geo-fence in which legal fishing vessels are fitted with a radio beacon for satellite monitoring of locations and monitored from the Police Maritime Unit at Betio, Tarawa; 6) trained Kiribati fisheries observers on board these legal vessels; 6) banning PIPA island landings to all but essential work; and 7) legal visitors to comply with landing protocols, with permits, and have PIPA staff present.

BIOSECURITY GUIDELINES

Summary of risks and needs

Biosecurity guidelines under development include comprehensive quarantine, surveillance, and response measures based on the risk assessments and summarized in Table 2. The level of risk in the Table (very high, high, and moderate) refers to the perceived likelihood of an invasion. No differentiation is made between impacts of different invasive species as they are all impacting and full implications are still unknown for some, e.g., different ant species.

The biosecurity guidelines being developed will provide a series of prescriptive tasks and data sheets that are intended to help guide the people responsible for the biosecurity actions identified in Table 2.

Proposed quarantine tasks

Because the PIPA islands are largely uninhabited and seldom visited, any invasive alien species (IAS) incursions could remain undetected for long periods and become expensive or impossible to eliminate (e.g., in the case of invasive ants). The emphasis therefore needs to be on invasion prevention. The highest priority needs include: 1) effective vessel quarantine together with IAS control at the ports of embarkation and arrival, e.g., Betio (Tarawa), Kiritimati and Kanton as part of the certification process under the pending Biosecurity Act; and 2) building on existing Agriculture Division process, including datasheets and reporting. The most urgent tasks in support of this process are to remove rats from inter-island freighters and this is starting to be implemented by Quarantine staff, initially at Kiritimati, and will be extended to Tarawa (and subsequently Kanton), using combinations of permanent bait stations and traps on the vessels and searching for rodent sign, and having independent verification via Government staff and passengers. Because of limited staff and potential work bottlenecks, collaborations between Quarantine and Environment divisions of MELAD along with port authority staff are essential in order to achieve effective results and these are being formally established, initially at Kiritimati. Future timetabled needs for freighters include surveillance for other IAS on vessels and at the ports, to include invasive ants, weeds, and birds.

Quarantine of fisheries vessels could be approached in a similar way with certification of pest-free status being verified by trained observers present on the licensed vessels at departure and throughout the fishing voyages. To date, the observers have been trained in a fisheries role only but they will be retrained to include IAS responsibilities. All other visiting vessels, e.g., research and management vessels are required to adhere to biosecurity guidelines as part of the permitting process or have their own approved biosecurity plan in the case of landing parties.

A key need at Kanton is to have quarantine representation on that atoll to ensure local quarantine procedures are strictly followed. This need increases further with future IAS eradications proposed for the atoll and increased ecotourism which might also see the reopening of Kanton Airport.

A recommended timeframe for key quarantine actions is:

2010 - begin rodent control on cargo vessels (Ag, underway)

2011 - begin rodent control in port compounds at Tarawa and Kiritimati (Ag); verify effectiveness of cargo vessel work (Ag and independent)

2011 - train fisheries trainers in biosecurity for them to train observers in rodent surveillance and control, but also awareness of other IAS (Independent/Ag)

2012 - other IAS in port compounds – survey/ surveillance, review/refine training of fisheries observers (Ag)

2012/13 - aim for Kanton Quarantine officer by now (MELAD).

Proposed surveillance tasks

Although quarantine is the key need, surveillance of priority islands is still advisable in order to detect pests before they become fully established and/or impact severely on sensitive biota. This will be addressed via Government observers present on all licensed vessels visiting the PIPA islands whether they are undertaking patrols, research, management or tourism. Guidelines are being developed to monitor sensitive indicator species, e.g., blue noddy (*Procelsterna cerulea*), and search for pest sign including, direct observations, gnaw-marks on eggs and discarded bird bones. These data will be held by the PIPA office.

In the case of the now pest-free islands (Rawaki and McKean), landing is generally discouraged to minimise risks of unforeseen incidents (IAS and accidents) and to set an example for all to follow. The exception would be if the government observers and other technical people present on vessels offshore believe there may be problems ashore. For example, if observers see sign of illegal landings on pest free islands and/or note that the sensitive indicator species are scarce, there is a standardized checklist for each observer to follow (Table 3).

Although the key need is to develop quarantine procedures to prevent incursions, there will always be some risk of pests reinvading. The biosecurity guidelines being developed for the PIPA do include recommended responses to invasions, including the broad approaches in Table 4.

Table 3 Example of a step by step approach for surveillance of pest-free PIPA islands.

Step	Activity	Items needed
1. Rawaki and McKean – pest-free islands teeming with birds		
1	From offshore, scan the entire foreshore for signs of illegal landings, shipwrecks, and, if it is possible to get in close enough, any sign of cats/rats on the upper beaches. From the vessel do a fly-on bird count – in evening (5.00 pm to dark) anchor boat at safe site c.100-150 m out from "the landing" and count the small sensitive birds (blue noddy, shearwaters and storm-petrels) flying to shore and within 100 m of your boat, i.e. a 200 m wide swath. **If bird counts are high on Rawaki and nothing suspicious seen, then no further work is required except to complete the survey form. If fly-on counts of blue noddies at Rawaki are < 50 and/or there is sign of landing or other suspicious sign at either island go to step 2**	Binoculars, surveillance form, instructions for fly-on counts
2	If you suspect there is a problem on the island and landing conditions are OK, follow biosecurity landing protocols and go ashore to search for invasives and their sign particularly focusing on: - tern/noddy colonies - are there any rat-eaten egg-shells or gnaw marks on any bird bones? - are there any ants on eggs or chicks or at the landing sites/structures? **If invasive sign is found on eggs or birds photograph and go to step 3 (rodents) or 4 (ants)**	Landing permission, landing protocols, safe landing gear, vials with preservative, digital camera, survey form, map of island, detailed methodology
3.	From late afternoon search for rats and other vertebrate predators into the night, and estimate numbers seen and map where they were seen and map where you have been. If rats are more extensively spread and there is not enough bait at hand (5 kg/ha required) to cover the island, do not attempt to poison them. Instead confirm species by catching and collecting several individuals by running them down (easy to do during the day) and weigh and measure and collect specimen as per data sheet. **If rats or other IAS are found alert the PIPA office immediately (Tukabu Teroroko ph +686 29762, mobile +686 94571) and provide details as more information may be needed. Tukabu will contact members of Biosecurity Committee for further advice.** **The boat should remain near island (in case more information is needed) until cleared by PIPA office to leave**	Strong headlamps or torches, batteries, ruler or callipers, 300 g Pesola balance, specimen jars and ethanol preservative or freezer. Pestoff bait (brodifacoum) - ideally have 100 kg available on patrol boat.
4.	Other surveillance If invasive ants are found at seabird colonies, determine their distribution on the island by establishing standard ant survey stations If invasive plant species (e.g., lantana, *Pluchea*) are found, photograph, determine the location of these sites by GPS and mark on a map of the island. If there are few plants, remove all the plants by digging them out taking care to include the entire root system as well as all seeds and place all these in a sealable container for later incineration. Also mark the sites on the ground with coral cairns in order to check for re-growth on later visits. **Go to step 5**	Ant survey kit containing vials, sugar solution, protein lures, preservative, marking pens, GPS.
5	All surveillance data and reports to be sent to PIPA office for follow-up action and filing	Weed surveillance booklets, camera, spade, containers, map of islands, data sheet, GPS.

Note that Enderbury and Birnie will be added to this island grouping once rats are removed – currently these and all other islands should be checked for signs of illegal landing, wrecks, etc.

Can planned biosecurity implementation work?

Action is urgently being directed towards the most likely pathway (cargo and fishing vessels) that could bring additional invasives to the PIPA and is based on the priority setting of Table 2. These actions include the use of rodent bait stations with brodifacoum, which has a fast kill rate, and rodent kill traps. This will be complemented with rodent control at the departure ports, mainly Betio/Tarawa and Kiritimati, and also at Kanton. When these most urgent procedures are working effectively, as determined by independent audit, vessel surveillance will be extended to incorporate searches for invasive ants, other invertebrates, reptiles, and weed seeds, including addressing IAS control at the port compounds and other nearby sources of IAS.

The success or otherwise of these proposals depends on sustained commitment in key areas including:

Developing trust and effective working relationships amongst government staff and with captains of fishing vessels, freight vessels, tourist vessels, and other vessels

Cooperation of community as passengers on vessels, and visitors to and neighbours of the port compounds

Having capacity and tools to do an effective quarantine job at source ports

All breaches of protocols and related issues are reported for court proceedings

Having the ability to respond effectively to biosecurity issues, e.g., mobilising surveillance aircraft and vessels, including patrol boats, to intercept illegal vessels

Having effective pest surveillance and an ability to respond quickly to any invasives arriving at the PIPA

Table 4 Summary of emergency response needs for the PIPA.

Objective	Tasks and responsibility
Identify biosecurity advisory team	An interim team led by PIPA Director has been identified to provide advice - available by phone if needed (PIPA)
Confirm identity of invading IAS species	Species-specific approaches e.g., for rodents capture by running down, trapping, sticky pads for hair; for specimens photograph, measure head and body length, also tail length, preserve in freezer or preservative; Collect and preserve any ants that appear potentially IAS; Collect weeds in sealed bags; GPS sites; Describe size and coat pattern of cats (PIPA Director/GoK rep)
Consider feasibility of immediate eradication with advisory team	With advisory team's phone advice via PIPA Director , assess whether IAS may be able to be eradicated immediately – e.g., cats by shooting and/or running down in the open; weeds by bagging, GPS site. (PIPA Director/advisory team)
Response procedures known	Broad response procedures for most likely invasives are being developed; include response team, bait etc availability, transport, timing of response and minimising impacts on non-targets (PIPA Director/Response team)
Funding	Emergency funding sources are currently an issue, but will be less so as the PIPA Trust undertakes fundraising (PIPA Director)

Improved quarantine procedures for all international vessels operating in the PIPA

A budget to cover all aspects of equipment, personnel, training, and emergency responses. Some of these costs can be passed on to PIPA users but an ongoing internal budget is required

A Biosecurity Team that includes individuals experienced in managing quarantine, surveillance and response issues

Implementing biosecurity education for targeted groups and the community.

Each of these steps is needed in order to sustain the island restoration gains through pest removal that are currently being made in the PIPA via pest removal. Sustaining this level of biosecurity commitment may at first seem expensive and daunting, especially to Kiribati staff. All of the above needs are ultimately achievable, but biosecurity implementation should begin with high priority needs first, i.e. addressing rodents on cargo vessels as is currently the focus, followed by fishing vessels and ports. Gradually, surveillance and control of the other IAS that can threaten the PIPA should be brought in after this together with increased education. Currently some establishment costs of biosecurity are being met partly by aid projects including CEPF- and NZODA-funded work, but in future the costs of sustaining effective biosecurity needs to be borne by biosecurity users, i.e. revenue generated from the fisheries licenses, freighters and research/tourism expeditions. Much generic IAS material is also widely applicable to the PIPA and Kiribati generally, including technical and education material (e.g., ACP 2010, PII 2010 draft, Tye 2009, Veitch and Clout 2002).

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Bagging them all in one go – personal reflections of a project manager about community based multi species animal pest eradication programmes in New Zealand

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Abstract Animal pest eradication programmes conducted by government agencies on offshore islands have encouraged an increasing number of community groups to attempt similar projects on islands as well as the mainland. Community groups often have limited resources but balance these with inspirational generosity and many thousands of volunteer hours. I have learnt nine key lessons regarding community-driven eradication campaigns. These lessons include the need to: know the target species and its environment; produce detailed but simple plans; have a network of contacts; foster a support network of experts; develop a toolbox of techniques; understand the difficulties of detecting pests at low densities; and value the roles of people who want to improve their local environment. Taking these lessons into consideration should assist in the successful planning of future community projects of a similar nature.

Keywords: Community partnerships, limited resources, efficient management systems, increasing experience, learning by doing, continuous improvement

INTRODUCTION

Animal pest eradication programmes in New Zealand were once only undertaken by the Department of Conservation (DOC) on isolated offshore islands (e.g., Bellingham *et al*. 2010). Successful programmes, combined with increasing public concern over the continued decline of our native species, have encouraged increasing numbers of community groups to take up the challenge of attempting animal pest eradications on the mainland. When undertaken by community groups these projects often involve very limited resources countered by inspirational generosity, which is demonstrated by the many thousands of volunteer hours that are expended. This paper summarises some of the challenges and lessons learned from personal experiences with community-based multi species animal pest eradication programmes. I identify and discuss nine key areas within a "learning by doing" approach using examples from mainland and island eradication projects in New Zealand.

LESSONS

Lesson 1: Know thine enemy and its territory.

The first rule of engagement is to know what you are dealing with and where it lives. This knowledge is required to determine whether an eradication is possible and how much it will cost. Importantly, this information needs to inform the client. People involved need to know what they are getting themselves into; a realistic view of what will be required to do this work is essential right at the start.

It is also necessary to demonstrate that there is a good reason to undertake the project. In New Zealand, eradications of invasive species are generally undertaken to protect endangered native species and/or threatened environments or to provide an environment free of animal pests as a refuge for native species.

Making this knowledge available does not necessarily require a large investment in monitoring to determine numbers of each pest species present. However, it does require knowledge about the pest species present, the effects they have on native species, and how introduced species interact with each other as well as with native species. This latter point is important because there may be prey switching or other imbalances if a predator species such as cats (*Felis catus*) is removed but their prey species, which might be rabbits (*Oryctolagus cuniculus*), are to remain.

Knowledge of local behavioural ecology is important. For example, rats are more likely to swim in summer months at Great Barrier in the northern Hauraki Gulf, most likely due to increased competition for food and dispersal of juveniles in search of their own territories. Other local environmental factors may affect eradication operations such as terrain and vegetation cover. An early understanding of potential issues that may affect the success of an eradication is vital so that sufficient time is available to plan solutions.

Similar projects conducted elsewhere can assist with planning. These may reveal issues encountered with the targeted species, non-target species, and the project environment. For example, eradication monitoring after the spread of baits within the fenced Maungatautari Ecological Island project indicated that mice may have persisted in windrows of logs and vegetation, feeding on the seeds of weedy vegetation inside the fence. Both factors may have resulted in some mice not eating bait.

At Tawharanui Regional Park, north of Auckland, livestock needed to be removed before aerial bait drops. The stock were also needed to keep grass short enough before the drops so that pasture did not provide food or shelter for rodents or restrict access to bait (Ritchie 2002). After the eradication attempt, mice were detected in long rank grass that had been retired from grazing many years before. The lesson here is that short grass is important to reduce mouse habitat and increase accessibility of bait to mice. In hindsight we should have talked to more people and considered this possibility more carefully.

Fig. 1 Rotokare Scenic Reserve – Taranaki.

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At Rotokare Scenic Reserve in Taranaki (Fig. 1), knowledge from the above two projects saw felled tree material during fence construction collected and stacked into windrows outside the fenced area. A large mob of sheep was also run inside the fenced area before the aerial bait drops so long grass was grazed to almost bare ground (Ritchie and Prankerd 2007). Although too early to be sure (the eradication only commenced in 2007), indications are that no mice survived the eradication operation. Unfortunately mice were detected and subsequently caught in late 2009; likely due to a maize truck entering to access a property on the other side of the reserve. The message here is: know your territory and plan for the inevitable if you can't plan against it.

Lesson 2: Plan the work – work the plan

Careful planning is essential. Some people find the process of developing and writing planning documents frustrating and a diversion of resources away from doing 'on the ground' work. Nonetheless, the work must be meticulously planned if it is to successfully deliver on project goals. Resources are always tight for community based projects; funding sources are limited and highly competitive. These are all strong reasons why credibility has to be demonstrated and methodical project management outlined through good planning.

I use the KISS principle: "keep it simple stupid". This may sound derogatory but it highlights that simple plans with easy language and clearly set out timelines and processes have the greatest chance of acceptance by stakeholders.

However, avoid mountains of paperwork – quality is more important than quantity. Key planning elements for the project need to be carefully defined. It may be necessary to select the best candidate out of a number of proposed projects. Stakeholders such as the community, clients, and funders may need convincing that the project is feasible.

The project manager needs to consider what would happen should they become personally unable to continue with the project. A measure of the quality of plans is whether another person with a reasonable level of skill could take over.

Each project will need an operational plan that clearly sets out how the work will be done (e.g., Prankerd 2007). This is the key document for people doing the work on the ground. Because eradications require rapid responses to new issues, operational plans need to be living working documents. They do not sit on shelves and gather dust, they need to be coffee stained, flecked with dirt and a bit torn because they are reviewed, implemented and amended constantly along the path to eradication.

Documentation is also critical; often there is a lot to think about during these projects. It is important to document how tasks were conducted in order to track progress but also to help others who follow with similar projects. Apply the KISS principle; build simple systems into operational planning and don't over-complicate recording systems. Consider the use of graphic techniques such as GIS (Fig. 2), station diaries and simple recording forms.

Lesson 3: Eradication is done once

Do it once and do it properly because it can be hard to rebuild confidence to repeat an eradication that fails. It can be very difficult to convince stakeholders and funders that you: a) know why the eradication failed, b) have measures to prevent it happening again and c) be able to convince people that these measures will work. Never ever compromise on quality – apply this to all aspects of the project including planning, community consultation, people, gear and equipment. Some of this may cost more but consider that the short term cost of success far outweighs the ultimate cost of failure.

Lesson 4: Manage expectations carefully

Community groups often do not realize that after an eradication considerable work may be required to sustain a pest free area over the long term. For example, there is a perception that you can build a pest fence, do a bait drop and then the job is complete. In fact, all multi-species pest eradications on the mainland have required a considerable amount of ground work to remove pests remaining post aerial drops and protect against reinvasions. To my knowledge Rotokare Scenic Reserve is the only one of these areas that is pest free. A number of others are tantalisingly close and may well get there very soon but will continue to require ongoing efforts to maintain this status.

Stakeholders need to know what eradication means. There is often confusion between control and eradication. Control means some invasive animals will remain. Eradication has a zero tolerance policy; often 90% of the effort goes into getting rid of the last animal. This can be very difficult and requires incredibly hard working and dedicated people, often assisted by some very smart dogs.

It is important to sustain the effort, to be realistic about how long and what will be required to reach completion. I use continuous review and improvement and always support, listen to and nurture those people who are out there doing the work in the field, often in physically demanding and monotonous conditions.

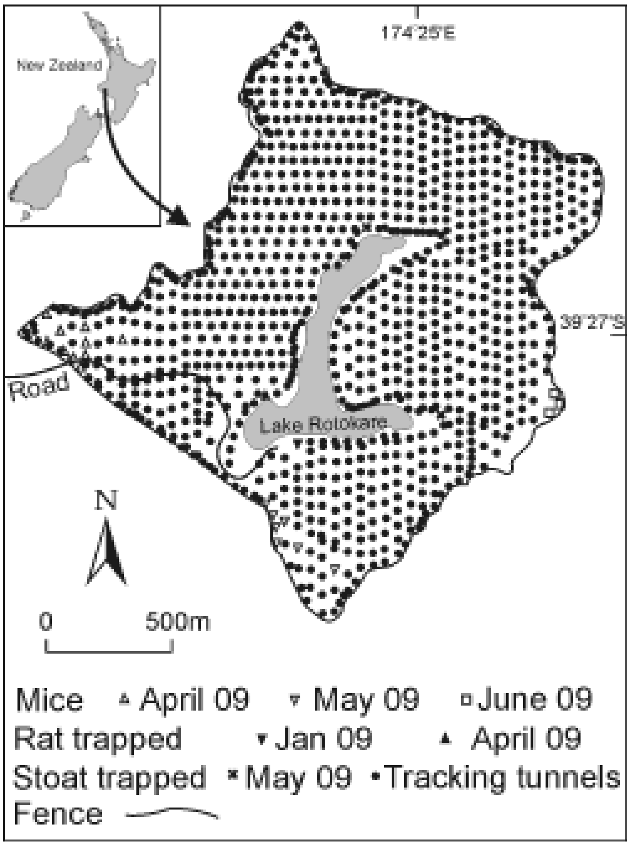

Fig. 2 GIS based mapping system used at Lake Rotokare to map and plan responses.

Lesson 5: Build a network of contacts

Despite some claims to the contrary, there are no experts in eradication work. Each eradication provides new lessons. I find a network of contacts invaluable. They help me in all aspects of eradication work and include a wide cross section of skills; bait manufacturers, animal pest ecologists, helicopter pilots, hunters, editors, field people, public and community relations people, iwi Maori advisers, and my husband. I can access these people whenever I need to and they always help or if they can't, they know someone else who can. Often I will call upon a few at a time. It is always important to make sure you thank them and acknowledge their contribution.

In one example, rats were nowhere to be found for about six months after aerial baiting at Kaikoura Island. When they were detected again we had to regroup and consider our next plan of attack. To aid my understanding, and give the Motu Kaikoura Trust and its hard working ranger on the island confidence that my advice was as good as possible, I contacted a range of people including DOC island specialists, rodent ecologists at Landcare Research, the bait manufacturer, and a DNA authority at the University of Auckland. These people were invaluable and together we formulated, and continue to refine, a detection, response and prevention programme for the island.

Lesson 6: Build a support network

The network of contacts is also invaluable for moral support. Despite the best intentions a lot of personal energy and commitment can be invested into eradication projects.

Challenges can arise such as when a constant and seemingly unstoppable stream of rats was swimming to Kaikoura Island from Great Barrier Island, mice arrived on a maize truck at Rotokare or the barge got delayed when taking 18 tonnes of bait to an island. That is the time when I call the network, ask them to help me stay sane and reassure me that my responses are best solution (Ritchie *et al*. 2009). This is particularly important for people working on community based projects for whom many of these tasks are new and very daunting.

Lesson 7: Develop a toolbox of techniques

Just as a good builder rarely goes to a job without a trusty belt pouch filled with essential tools, so too is it rare that a multi-species eradication can be completed with just one technique or tool. Different tools are often required for different species. Even for the same species, a range of tools may increase the chances of achieving eradication because getting that last animal may require novel approaches. For example, at Tawharanui, despite tracking tunnels with peanut butter and rabbit meat as lures, some rats bypassed them along fencelines but were then captured in traps. At Rotokare, despite a 50 x 50 m tracking tunnel grid (about 1100 tunnels, Fig. 2) two stoats escaped detection until they were caught in traps.

Quality must always reign over quantity. Poorly set traps, a bad shot with a rifle, or rotten bait can result in bad experiences and increase the difficulty of catching some animals.

It is also important to know how each tool works. For example, there is a common misconception that tracking tunnels measure density when they only measure presence. One busy mouse (*Mus musculus*) can cover a tracking card with footprints. Other issues are with rat traps, which may not always be sensitive enough to catch mice, and some toxins, which are less effective than others and for which inappropriate use can result in bait aversion. These issues must be considered if an eradication programme is to avoid costly mistakes.

There is no need to reinvent the wheel. The network of contacts can help with knowledge about available techniques, their efficacy with specific pests, and situations where they work best. Other projects are a knowledge source that can be learned from and adapted to the current situation. To reciprocate, I in turn provide knowledge and experience to others.

Lesson 8: Absence of evidence is not evidence of absence

Detecting animal pests when at low densities is difficult, especially for animals like mice with very small home ranges. Unless funds are unlimited or the project is very small, it is often not feasible to set up the high density of tracking devices required. For example, monitoring during eradications conducted in the fenced cells at Maungatautari Ecological Island (www.maungatrust.org*)* found that all mice were detected using a 50 x 50m tracking grid. This may in part be due to behavioural changes by mice in low densities, when their home ranges can become measured in hectares. The Maungatautari work also found that between about October to March it was very hard to detect anything, due to abundant natural food. However, this may also be due to the fact that invertebrates quickly find bait in tunnels and reduce its attractiveness.

The point here is that absence of evidence is not evidence of absence. Intensive monitoring may be required with a range of devices over at least 2-3 years. In New Zealand, this covers all seasons and levels of detectability at least twice, which increases confidence that the last individuals were actually eliminated.

Patience is a virtue in these projects but may be hard to impress on community groups. However, we managed to do so at Rotokare where the Trust has so far resisted declaring the area pest free. They have also resisted reintroducing lost native species until late 2010 because declaration of pest free status and/or reintroduce native species too early may compromise the whole project. It is difficult to recover stakeholder confidence if pests are detected after they are assumed to have been eradicated. Furthermore, if native species are reintroduced that are sensitive to pest removal tools such as toxins and traps, the chances to quickly and effectively remove a newly detected pest may be compromised.

A lesson we have all learned doing eradication projects, especially those behind pest fences, is that pest free status may only be temporary until a cyclone or once in a lifetime thunderstorm breaches the fence. In these circumstances, an alternative approach to continuously chasing the last animal is that some may be tolerable if kept at biologically insignificant levels. In such cases, effective surveillance and pre-developed response strategies may be all that is required, although these responses need to avoid damage to native species.

Lesson 9: He tangata He tangata He tangata

The people, the people, the people. I'm going out on a limb here – don't we do this work for ourselves? We believe implicitly that what we are doing is the right thing to do. We are saving native species and ecosystems, empowering communities, and demonstrating that people can make a difference. Community-run eradication projects are all about people; mainly people who want some help to make a difference in their local environment.

Fig. 3 Proximity of mussel farms to Kaikoura Island.

However, there will also be some people who oppose an eradication, whether as a genuine concern over the use of toxins, the cost of a project, or a concern that it will upset their personal freedoms, e.g., no deer to hunt after they have been eradicated. If the concerns of such individuals are considered, the planning process becomes more robust. Often the information obtained by opposing groups has been misconstrued or is incomplete and it is possible to reach a compromise. For example, at Kaikoura Island mussel farmers (Fig. 3) were initially not prepared to support the aerial baiting operation because of concern about possible impacts on shellfish. They were genuinely concerned about how the use of brodifacoum in close proximity to their farms might be perceived and also about the potential effects of brodifacoum poisoning. We met with the farmers and collated technical information, including the results of an accidental spill of 18 tonnes of bait containing brodifacoum into the sea off the South Island. This data was sent by the farmers to be independently analysed at a science laboratory.

The outcome was support to do the drops conditional on: testing mussels before and after the drops; liability insurance taken out by the Motu Kaikoura Trust; conditions relating to undertaking the drops outside the harvesting season; and how we would fly the area immediately adjacent to the farms (Ritchie 2008).

Fig. 4 Bait breakdown monitoring cage.

Fig. 5 Aerial baiting coverage at Kaikoura Island.

The Rotokare project was also a challenge. A 230 hectare forest remnant with a lake in the middle surrounded by a pest-proof fence may seem small and easy. However, it also had public access in summer for boating, picnicking and walking, lambing on the surrounding properties in winter when aerial baiting took place, and 12 species of animal pests ranging from mice to goats (*Capra hircus*). We went through about six versions of the operational plan (Ritchie and Prankerd 2007) making changes as we gained information and more people read it. The farmers helped write the conditions for the aerial baiting contract, which required all activities to be within the fence and there were observers watching for bait going over the boundary on baiting days. Being flexible and open minded is the key when planning eradications.

Communication is the key. People need time to think about discussions and also need to feel that their opinions have been treated with respect. A common language is required with information presented in a form that suits the audience. Always serve up the good with the bad e.g., there can be adverse effects with some toxins but balance this with the advantages and gains. If both sides of the story are not presented some people may encounter contrary information then use it as evidence that information is being hidden. It is also necessary to be honest if answers are unavailable. For the Tawharanui project (Fig. 4) we didn't know how long it took for Pestoff 20R (a brodifacoum based bait) to break down in the environment or what a livestock withholding period should be so we undertook studies to find out.

There are many misconceptions about the aerial spread of toxic baits with helicopters and these have been repeated with every eradication project I have worked on. Consent authorities often permit aerial baiting under aerial spraying sections of regional and district plans (unlike aerial spraying where there can be drift, there is no drift with aerial baiting). Another key issue is the public perception that bait is applied in an uncontrolled fashion and that much of the bait goes into the sea during operations that involve coastlines.

Such issues should be approached pragmatically and head on. Local authority planners are always open to new information but need to be satisfied that the information you provide can be substantiated. It is important to aid their understanding.

Fig. 6 Return of kiwi to Tawharanui Open Sanctuary.

Changing public perception is difficult but not impossible. It helps to view problems as challenges to be overcome. At Kaikoura Island, we did this by inviting some people who were concerned about the aerial baiting operation (Fig. 5) to observe one. They met the pilot, had a lesson on how our monitoring systems worked (random bait grids, bucket flow checks, GIS downloading of flight lines after each load) and went out in a boat to watch baiting on coastal cliffs. The latter included going ashore to look for and count baits on exposed coastal reef platforms. The result was an appreciation of the rigor employed during these operations and reduced concern.

People are always vital components of the projects I work on. They inspire and provide invaluable assistance in many ways including the championing of projects and the undertaking of the work on the ground. Often this work is voluntary and requires considerable time, effort, and cost to each individual. This is inspirational generosity – these people often repeatedly assist and then find others to expand the pool of helpers. For example, at Glenfern Sanctuary, a 230 ha pest fenced peninsula on Great Barrier (www.glenfern.org.nz), a 50 x 50 m tracking tunnel grid has been installed after the aerial spread of bait in winter 2008. Monitoring this grid monthly is hard monotonous work but it is managed by highly capable local people and a band of volunteers from all over New Zealand. Many volunteers return repeatedly to walk in steep bush placing ink cards and bait into >1100 tunnels. The same is the case at Rotokare. The knowledge these people build up should not be undervalued.

We encourage these participants to write notes in project diaries of any ideas, observations they might have. Acknowledging these efforts is essential. We do this by newsletter updates, barbecues, celebrations, and invitations to special events. One such event at Tawharanui was the release of North Island brown kiwi (*Apteryx mantelli*) in 2006 after a 50 year absence from the Auckland mainland (Fig. 6). Two hundred and fifty people came to the first release on a wet, wild day. But it was one way to celebrate, encourage and reward these workers and their community.

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I dedicate this paper to Tony Bouzaid, the driving force behind Glenfern Sanctuary.

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Increasing the return on investments in island restoration

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Abstract The effects of invasive species are now being reversed through successful eradications of unwanted organisms on increasingly large and remote islands. Although these successes represent an encouraging trend, we suggest that it is time to examine their pace and scale, given current advances in the science and practice of eradication. To date, most eradications have been implemented as "one-off" projects, with little coordination across islands. As a consequence, opportunities are missed to achieve economies of scale in planning, permitting, staffing, and purchasing that could lead to a considerable increase in time- and cost-efficiency in reaching eradication goals. A more coordinated cross-island effort could also allow for greater development and retention of specialised capacity, which would not only further enhance the efficiency, but also reduce risks of failure inherent to eradication programmes. More funding is needed to support eradication efforts. Should these funds become available, a programmatic and coordinated approach to their use could greatly increase the outcomes achieved. This coordination could include multiple invasive taxa and/or multiple islands that are managed in strategic sequence. By developing and supporting a planned sequence of projects, e.g., for archipelagos or regional clusters of islands, eradication efforts could be designed to achieve efficiencies in planning and implementation that could result in greater return on investment than an island-by-island approach. A regionally- or internationally-supported systematic initiative could also help overcome a major limiting factor in island restoration: insufficient in-country capacity to support a sustained eradication programme. A ship-based platform may be a highly effective tool to implement this more programmatic approach; for example, it could help overcome obstacles to implementation on remote and/or small and/or inaccessible islands.

Keywords: Economy of scale, efficiency, eradication, invasive species, planning

INTRODUCTION

Invasive alien species are a key threat to native biodiversity (Vitousek *et al*. 1997), particularly on islands (Mulungoy *et al*. 2006). Fortunately, invasive species are increasingly being eradicated from islands as planning and technical tools improve (Parkes and Panetta 2009). Unlike continents, islands can be more easily defended from new invasive species by good quarantine and border security (Jarrad *et al*. 2011; Russell *et al*. 2008). If eradication is achieved, unlike sustained control, threats are entirely removed, which maximises benefits to native species and ecosystems. The relative cost/benefit ratios of eradication can be better than those for sustained control (Panzacchi *et al*. 2007), although there are very few adequate analyses of these comparisons for protection of non-market values (Hone 2007). Furthermore, sustaining control, and the budget to support it, is very difficult for funding agencies (Parkes and Murphy 2003). Eradication does not require such long-term commitments, and there are many examples where eradication of a pest has resulted in major improvements of native biodiversity (e.g., Rauzon 2007; Rodrigues 2006).

Perceptions of eradications have also shifted from 'too hard' in the 1970s for views about rodents (e.g., Wodzicki 1978) to one of 'can do' due to successes for such diverse species of mammals as rodents (Howald *et al*. 2007), goats (*Capra hircus*) (Campbell and Donlan 2005), cats (*Felis catus*) (Nogales *et al*. 2004), pigs (*Sus scrofa*) (Cruz *et al*. 2005), and other species (Parkes and Panetta 2009) (see also the Global Island Invasive Vertebrate Eradication Database at www.islandconservation.org/db). The future still holds challenges. Some invasive species, or groups of species, remain intractable or difficult to eradicate either due to a lack of effective management tools as the case for *Suncus murinus* (Varnham *et al*. 2002) and most amphibians (Campbell and Kraus 2002), or because of life histories and behaviours that make it difficult to place all individuals at risk (e.g., most birds, invertebrates, weeds). Invasive species in aquatic habitats are often intractable because we lack suitable tools, they occupy habitats inaccessible to managers, and because aquatic species often produce vast numbers of cryptic, mobile dispersal stages. Eradication failure rates for species such as mice (*Mus musculus*) remain frustratingly high, often for reasons that remain unclear (Howald *et al*. 2007; Mackay *et al*. 2007). It is also unclear whether dealing with invasive species on large islands is just a matter of scaling up what works on small islands or whether new strategies and tactics will have to be developed (Parkes and Panetta 2009; Parkes *et al*. 2008). Nevertheless, accumulating successes have led to growing national (e.g., Aguirre-Muñoz *et al*. 2008; Anon 2009) and international (e.g., Genovesi and Shine 2004) interest in the role of eradication of invasive species as part of island restoration.

To date, only a fraction of the thousands of islands with invasive species have received management action. Reasons for this include the relative novelty of eradication methods, the inaccessibility, remoteness or large size of islands, and limits on the capacity of managers to engage beyond islands in their charge. As a consequence, eradication efforts are often *ad hoc*, planned and executed as "one off" efforts, driven by the presence of a local champion or proponents, focused on one pest species at a time, and on one island at a time. The economic and opportunity costs of this approach may be significant. If there are multiple pests on an island, there may be economies of scale in addressing them comprehensively while the eradication infrastructure is in place (Morrison 2007). Also, if island projects could be lined up in a strategic sequence, eradication activities among the islands could be sequenced efficiently, and the accrued expertise and experience of the eradication team could be retained.

In this paper, we argue that with advances in the strategies and tactics of eradication of invasive species on islands, it is time to ask how to increase the pace and scale of these achievements. Of course, one means of increasing the rate of eradications is to increase funding. We underscore the importance of increased private and public investment

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in this proven and timely conservation approach. But in addition to more funding, we may be able to increase returns on the available funds by investing in more programmatic and systematic efforts. With this investment one could develop a pipeline of projects planned and implemented in strategic sequence, using infrastructure and capacity across multiple island systems and international borders.

OPTIMISING INVESTMENT

Increased investment in pest eradication results in disproportionately large returns on island investment – even if it follows the single species, single island model. For example, the eradication of Norway rats (*Rattus norvegicus*) from Campbell Island (McClelland 2011) covered a much larger area than previously attempted and had benefits for many invertebrates and terrestrial and marine birds. Different proponents vary in their criteria for nominating one project over others, but because of the uniqueness and sensitivity of island ecosystems, they are usually underpinned by goals to protect biodiversity and, increasingly, to improve human health and livelihoods.

The trajectory of eradication successes might increase, however, if a systematic approach was designed, and funding was invested in its planning, infrastructure, and implementation. The incremental development of aerial spread methods against rats that led to the Campbell Island project demonstrates the value of such an approach (Towns and Broome 2003). Similarly, 'lining up the islands' and dealing with them as groups can: 1) reduce costs to assemble and apply the logistics required to conduct an eradication, 2) retain specialised skills in planning, delivering and monitoring eradication operations, and 3) improve the economies of scale and duration that would facilitate building community and local stakeholder support for proposed actions and anticipated outcomes. In some cases, local capacity building will be an important element. Experience has shown that community engagement and the facilitation of substantive stakeholder involvement can be crucial to success. In any event, ensuring stakeholder needs and perspectives are incorporated will be an essential part of the development of any regional or international proposal.

Several countries and regions are now prioritising islands for restoration, with examples in New Zealand, the Aleutians (USA), Mexico, the Caribbean, South Atlantic Territories (UK), and parts of the tropical Pacific. We believe that the next step could involve evaluating the benefits and strategies for implementing those priorities in a sequence designed explicitly to seek minimised programme costs, provide high quality eradication plans, satisfy the prerequisites for eradication, and achieve the biodiversity, economic and social goals set by stakeholders.

A MECHANISM – "THE GOOD SHIP RESTORATION"

Dealing with groups of islands in some planned sequence, especially oceanic groups or those in remote places, is constrained by logistics, including the transport of staff and equipment and their maintenance on site throughout projects. Where the lack of a suitable vessel and/or on-island facilities limits progress with eradication programmes, addressing this issue should perhaps be a priority for national and international partners.

A solution for logistic issues could be a fit-for-purpose ship. For example, a ship could be designed for use in the mid-Atlantic and deal with everything from reindeer (*Rangifer tarandus*) on South Georgia to mice on Gough to rabbits (*Oryctolagus cuniculus*) on Ascension islands. Such a vessel would be different from one required to sail round the Chagos Islands in the tropical Indian Ocean and deal with rats, or around Baja California and deal with suites of pests and weeds, or the Red Sea and deal with rats and goats (and pirates). Ships as a means of transporting the people and equipment required to eradicate pests from islands would be most appropriate where there is no shore-based infrastructure. Elsewhere, a ship may only be needed to provide transport and support for existing shore-based facilities.

NEXT STEPS

We propose that it is now time to discuss how to scale up these approaches to a global collaboration, and rigorously examine the economic merits of doing so. This should include an analysis of the economic feasibility and an assessment of the return on investment (relative to other options) of a ship-based approach using some specific island examples from different regions. A system for identifying and prioritising islands and archipelagos for restoration would also be needed (e.g., Donlan and Wilcox 2009). This might include assessments of the extent of regional or national interest in having particular islands or archipelagos included, relative biodiversity benefits, anticipated costs and local stakeholder engagement and "ownership". Where costs and benefits are about equal, projects offering the most local and national support should outrank those offering the least.

Once islands are prioritised, the specifications of vessels and infrastructure to support particular programmes would be defined and the availability of appropriate vessels and the costs of securing them (e.g., buying, leasing, chartering) could then be investigated. Our initial investigations indicate that many suitable vessels may be available for such programmes.

If these assessments were positive, agencies and individuals with interests and capacity to contribute could form a collective to develop and refine strategies and actions, to liaise with national and regional agencies, and to promote identified programmes to potential funders.

SOME SCENARIOS

We explored these ideas for three island groups and examined how they might benefit from a coordinated approach. Many other archipelagos, regions or sub-regions could have also been selected including:

- Equatorial islands in the Indian Ocean (Chagos, Maldives, Laccadives and Socotra) and other important seabird islands of the Red Sea.
- Eastern Indian Ocean chains of the Andaman and Nicobar Islands
- Southern Indian Ocean islands of South Africa and France
- Various island groups in the Caribbean
- Tierra del Fuego and associated islands
- South Atlantic Ocean islands from South Georgia and the Falklands/Malvinas north to the UK and Brazilian islands.

The following short list illustrates the range of physical and political constraints and opportunities that different island groups present.

Equatorial Pacific

Over 500 main islands and hundreds of smaller islands are situated within about 10 degrees of the equator in the central Pacific. The islands extend from Palau, the Federated States of Micronesia (FSM), the Republic of the Marshall Islands (RMI), Tuvalu, Nauru, Tokelau, the Northern Cooks to the Phoenix and Line Islands of Kiribati, and the Marquesas in the east. Most have one or more species of invasive animals as well as weeds of varying management difficulty. Most islands are populated but some are too remote or too small to support permanent human habitation.

Some eradication projects have been conducted in the area, including Demonstration Projects under the Pacific Invasives Initiative (www.issg.org/cii/PII). There has been some prioritisation of the biodiversity values on these islands through National Biodiversity Strategies and Action Plans, and of potential invasive species eradication projects. For example, 1402 potential eradication projects have been identified on 79 islands or groups of islands in Palau, FSM, and RMI and ranked to list the top 20 eradications (mostly of rats) to maximise biodiversity gains (Wegmann 2007). Seven of the eight islands in the Phoenix chain (Kiribati) were surveyed by Pierce *et al*. (2006) and PII subsequently coordinated the removal of *Rattus tanezumi* from McKean Island (49 ha) and *Oryctolagus cuniculus* from Rawaki Island (58 ha). A planned eradication of *Rattus exulans* from Birnie Island (48 ha) was not undertaken (Pierce *et al*. 2008). These eradications used a ship to transport people and equipment, were limited to small scale operations manageable without helicopters, and avoided long periods ashore. Eradication operations on larger islands in the chain (Enderbury and Orona are over 500 ha) and with rats and cats (the latter at least on Orona) would require more sophisticated infrastructure and more time. The operations undertaken were quite risky to the people involved and in terms of the narrow "window" of time in which suitable weather could be exploited. Nevertheless, the campaigns demonstrated that eradications on some of the most remote unpopulated islands in the world could be successfully undertaken with appropriate planning, a determination to succeed, and a vessel supporting the operation.

Western Mexico

There are about 300 islands off the Pacific coast of Mexico and in the Gulf of California. These islands are important biodiversity resources with high levels of endemism (Case *et al*. 2002). Mexican organisations have been successfully managing invasive species on some islands over the last decade (Aguirre-Muñoz *et al*. 2008). Recent rat eradications (Samaniego *et al*. 2009) relied on a combination of Mexican Navy ships and private helicopters. Key constraints have been a lack of reliable access or any suitable on-shore facilities on many of these unpopulated, arid islands. While the support of the Navy has been invaluable, they have other duties and cannot necessarily commit to fit in with a restoration project's needs and timing. A vessel dedicated to restoration programmes would allow the Mexicans to increase the rate of eradications and potentially begin some of the currently less feasible projects on some larger islands. These could include removing feral cats and goats from Espiritu Santo and Cerralvo, cats and mice from Guadalupe, sheep (*Ovis aries*) and cats from Socorro, and ungulates and rodents from the islands of the Tres Marías Group.

Tasmania

The island State of Tasmania is an important repository for many Australian species extirpated by introduced predators and herbivores such as the red fox (*Vulpes vulpes*) on the mainland. The State also includes about 300 smaller islands that are themselves important nesting sites for seabirds, as well as potential arks for sustaining species threatened on the main island of Tasmania – a threat that is increasing since foxes have arrived (Parkes and Anderson 2011), and Tasmanian devils (*Sarcophilus harrisii*) are dying from disease.

The Australian Federal Government has identified which invasive species are present on 56 Tasmanian islands (Terauds 2005), and indicated its intention to do something about them on these and all other Australian islands (e.g., for exotic rodents; Anon 2009), and prioritised these intentions for the top 100 islands of the thousands of islands in Australia (Ecosure 2009). The prioritisation listed 15 Tasmanian islands.

Some of these islands are easily accessed by boats or helicopters from the main island, but many are either remote (e.g., Macquarie Island) or off uninhabited coasts. A ship is required to access these islands and, perhaps, to support ship-based eradication operations.

CONCLUSIONS

Exciting advances in the past decade have led to increases in the number of invasive species targeted, the size of islands treated, the pace of developments and, the number of countries involved. Yet, constraints associated with a lack of continuity, capacity and funding remain significant impediments to further progress. Furthermore, eradications of pests on remote or inaccessible islands and in countries without extensive experience and capacity will require an 'industrial scale' response. We suggest that it is time to initiate a coordinated and progressive international programme to address these constraints and to maximise the return on investment from limited restoration budgets.

Our suggestion is to assess whether a more systematic and perhaps ship-based approach might achieve these goals. Like the *Calypso* and *MV Steve Irwin*, which are seen as symbols for marine conservation, a ship-based programme focused on island restoration could become both a practical tool and a symbol of cooperation and conservation – two imperatives for islands in this time of uncertain global change.

ACKNOWLEDGEMENTS

We are grateful to our many colleagues in conservation whose skills and determination have led to the successes that inspire and challenge us all to increase the pace and scale of this essential work. This paper has benefited greatly from the generosity of ideas and enthusiasm of many, and represents a milestone in an ongoing "global" dialogue. We look forward to the collaboration and restoration ahead.

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Creating an island sanctuary: a case study of a community-led conservation initiative

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Abstract The Pomona Island Charitable Trust is a community-led initiative with the vision of restoring Pomona Island to a pest-free state and maintaining it as an island sanctuary. The Trust aims to provide an accessible location for locals and visitors to see, hear and learn about the flora and fauna native to Fiordland. Since the Trust was formed in 2005, over 190 different volunteers have put in over 4700 hours of work on the island to remove five pest species: stoats, deer, possums, rats and mice. Over $165,000 of direct funding has been raised, largely from within the local Fiordland Community, with a further $130,000 of in-kind donations contributing to the work of the Trust. With all animal pests now removed from Pomona Island, volunteers have re-introduced South Island robins as the first of many planned translocations. Department of Conservation staff have described the Trust's achievements as "a model for community driven conservation". This paper presents a case study of the Pomona Island Charitable Trust. It focuses on the managerial initiatives undertaken to plan for the restoration of Pomona Island, the ways in which the Trust has worked with key stakeholders including the local community and the Department of Conservation, strategies for successful fund-raising, and maintaining momentum in a long-term community-led conservation project. Based on the experiences of the Trust, a model for successful community-led conservation projects is presented.

Keywords: Community conservation, island eradication, pest eradication, Fiordland

INTRODUCTION

Pomona Island (262 ha), within the Fiordland National Park, (Southwest New Zealand World Heritage Area) is the largest island in Lake Manapouri and is the largest inland island in New Zealand. Rising 340m above the lake, Pomona Island is a round-topped granite hill with steep sides, 500m from the mainland. Vegetation on the island is predominantly mixed beech-kamahi (*Weinmannia racemosa)* with rata (*Metrosideros umbellata*) and podocarp forest. Five pest species were present on Pomona Island: stoats (*Mustela erminea)*, possums (*Trichosurus vulpecula*), red deer (*Cervus elaphus*), ship rats (*Rattus rattus)* and mice (*Mus musculus*). These have had a major impact on the island's biodiversity and in particular its native birds. This paper outlines how a community-led project eradicated all introduced mammal pest species from the island.

In 1956, plans to raise Lake Manapouri by up to 30 metres for the generation of hydro-electricity were thwarted by environmental protests. Saving Manapouri has been described as New Zealand's first great conservation success story (Peat 1994). This paper outlines how the Pomona Island Charitable Trust is restoring the largest island in the lake to its natural state for the enjoyment of future generations.

In 2003, two local business people approached the Department of Conservation (DOC) about creating an island sanctuary on Pomona Island in Lake Manapouri and in 2005 the idea was adopted by some residents from the Manapouri township. Rough plans for eradicating stoats, deer and possums from the island were presented to DOC and these indicated the need for more formal eradication plans for each pest species. Following discussions between DOC and a few key local people, a charitable trust was considered to be the most effective means to manage the restoration of the island.

The Pomona Island Charitable Trust was incorporated in 2005 under the Charitable Trusts Act 1957. The Trust initially included seven founding Trustees and a DOC Advisory Trustee. Emeritus Professor Alan Mark, a world-renowned botanist, agreed to be the Trust's Patron. Since the Trust's inception, the number of Trustees has increased to nine. Each Trustee brings their own set of skills and experiences to the project and all are passionate about Fiordland. The Trust has three main Office holders: a Chair who is a farmer, with a good knowledge of local flora and fauna, a Treasurer who is a local business man and a Secretary with a marketing background. The other Trustees include an engineer, nature guides, helicopter pilot, tourism operator and local Maori. The Trust meets at least four times a year with regular email communication between meetings.

In 2006, a management agreement for ten years, with a right of renewal for ten years, formalised the relationship between the Trust and DOC. This agreement gives the Trust the autonomy to carry out a wide range of activities including research, pest eradication, species translocations, monitoring, advocacy and education.

The following factors have led to the success of this community-led conservation project (Fig. 1).

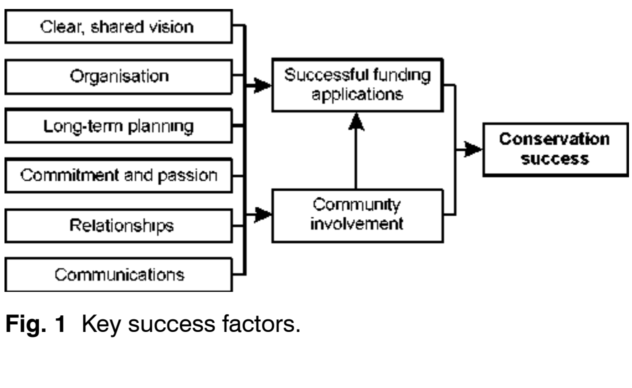

Fig. 1 Key success factors.

LONG-TERM PLANNING

Initial activities for the Trust included developing its vision and objectives, clearly defining its aims and developing plans to implement them. The vision of the Pomona Island Charitable Trust is: *to restore Pomona Island to a pest-free state and maintain it as an island sanctuary*.

Pages 496-499 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

The specific aims of the Trust are as follows:

Conservation: to eradicate all mammalian pest species from the island.

To ensure a high quality of indigenous biodiversity on the island in terms of both flora and fauna.

To reintroduce, through natural and assisted means, birdlife native to Fiordland within the Southwest New Zealand World Heritage Area.

To provide a safe habitat for endangered and threatened birds to breed thereby increasing the populations of individual species.

To monitor conservation activities and their impact on the island's biodiversity.

To transfer the experiences learned on Pomona Island to other inland islands in Fiordland.

Research and Education: to encourage research activities which enhance our knowledge and appreciation of restoration activities.

To promote awareness among both local people and visitors of the indigenous biodiversity potential of the island.

To provide an accessible location for people to see, hear and learn about the flora and fauna native to Fiordland.

Community Involvement: to ensure community involvement in the island restoration project through informal consultation and volunteer activities.

To promote the restoration of Pomona Island as something of which the local community can be proud.

Recreation and Tourism: to promote Pomona Island as a place for locals and tourists to visit and experience a part of Fiordland as it used to be.

These aims are supported by the following plans.

Communications Plan: good communications are essential for developing awareness and ownership of the project within the community, for fundraising initiatives, for keeping volunteers motivated and for getting good publicity. To develop successful communications the Trust identified key stakeholders, strategies for developing relationships with each of them and the communication method best suited for each group (Table 1).

As the project has progressed, the Trust has more clearly defined its local community target market with specific communications strategies developed for boat owners encouraging them to help the Trust keep Pomona Island pest-free.

Promotional messages consistent with the Trust's objectives have been: promoting the restoration of Pomona as being a project of which the local community can be proud; the creation of an island sanctuary where locals and visitors can see, hear and learn about the flora and fauna native to Fiordland.

Pest Management Plan: the management agreement between the Trust and DOC required the preparation of a professional pest management plan with eradication methods, costs and timescales for each species (Brown 2006). The plan was peer reviewed by the Department of Conservation's Island Eradication Advisory Group. A key component of the pest management plan was the involvement of volunteers at every stage of the restoration project. The work detailed in this plan has been completed (Shaw and Torr 2011)

Social Impact Assessment: prior to eradicating the pests from Pomona Island, the Trust decided that it would be beneficial to conduct a social impact assessment (SIA) (Shaw 2006). Most conservation projects are likely to have potential positive and negative effects (Cosslett *et al* 2004) and the aim of the SIA for the Pomona Island Charitable Trust was to identify and analyse the effects of the island restoration project on different groups and individuals in the local community. As Cosslett et al (2004) point out "failing to demonstrate the benefits of conservation initiatives to local communities may mean your work is less likely to be supported and may even be actively opposed by local people".

Benefits of a SIA for the Trust included: 1) promotion of community involvement in, and ownership of the project; 2) maximisation of positive outcomes; 3) the ability of the Trust to build on local knowledge and engage interested parties in the restoration of the island. As Taylor and Buckenham (2003) note, a project that invites participation from interested parties is likely to have a higher level of support and thus success. Consultation and partnership are seen as being important in pest eradication activities. 'Engaging in consultation, and being seen to engage' (Fraser 2006), can help local communities feel more involved in a project. The results from the social impact assessment fed directly into the operational plans for the eradication of each pest species from Pomona Island as well as into the Trust's communications plan.

Table 1 Key stakeholders and communications strategies.

Stakeholders	Communications Strategies
Local community	Local Media – Fiordland Focus, Fiordland Advocate, Southland Times, Otago Daily Times; Presentations to local groups; website; newsletter; Trust brochure; quarantine brochure; Art in the Park events
Department of Conservation	Meeting minutes; regular reports; face-to-face meetings
Media	Press releases; newsletter; invitations to key events
Environmental Groups	Media – Forest & Bird newsletter, reports; website; newsletter
Local iwi	Meeting minutes; reports; face-to-face meetings
Sponsors	Newsletter; website; reports
Visitors to Fiordland	Fiordland Focus; Trust brochure
Tourism Operators	Trust brochure; website; newsletter; local media
Researchers	Face-to-face meetings; website; reports

Quarantine Plan: Pomona Island is classified in the Fiordland National Park Management Plan as an "Open Sanctuary" Island which means that it is accessible to the public at all times. The preparation of a quarantine plan, following on from the eradication of all pest species, was essential if the Trust wanted to maintain the island as a pest-free sanctuary. The plan, prepared by a volunteer, aimed to minimise the risk of re-invasion of the five pest species that were originally present on the island and to prevent populations of the pest species from becoming established by catching every invading individual within a short time of their arrival (Willans 2007). Education is a key component of the quarantine plan. The Trust has worked hard with the local community, especially boat clubs, water taxi operators and individual boat owners to encourage them to make the necessary quarantine checks before they visit the island. Prominent signs are in place at key landing sites to remind boat owners of their responsibilities in helping the Trust maintain the island free of pests.

Restoration and Monitoring Plans: the restoration plan provides an overview of the pest eradication and a discussion of the species that the Trust would like to re-introduce to the island over the next five years (Shaw and Whitehead 2008). These include South Island robin (*Petroica australis*), mohua (*Mohouoa achrocephala*), saddleback (*Philesturnus carunculatus*) and kiwi (*Apteryx australis*). For each planned species translocation, volunteers prepare a formal translocation proposal and liaise with DOC staff. In order to assess the changes in flora and fauna on the island the Trust has put together a monitoring plan. This includes regular monitoring of vegetation plots and formal five-minute bird counts five times a year.

COMMITMENT AND PASSION

The Pomona Island Charitable Trust's success is due to the commitment and passion within the local community. Support from key sectors of the community ranged from the Mayor of Southland to the individual volunteers who put in the hard work on the island.

Three highly committed Trustees have shouldered the administration of the Trust and have also completed plans, implemented the eradication of pest species from Pomona Island and begun the re-introduction of bird species native to Fiordland. In addition, these same individuals have built relationships with key individuals, organisations and the wider community. Project management for these activities project has been mostly provided by the Secretary of the Trust on a voluntary basis with an estimated cost saving to the Trust of NZ$70,000.

The Trust maintains an email list of potential volunteers who are informed of volunteer work days (working bees). Since the first track was flagged on the island in April 2006, 193 volunteers have devoted nearly 4700 person-hours of work on the island. In the small communities of Manapouri and Te Anau approximately 10% of the local population has attended a working bee on the island. Many more companies and individuals have provided the Trust with financial support.

RELATIONSHIPS

Because Pomona Island is within the Fiordland National Park, the most important partnerships is with the management agency, DOC. The Trust and local community have been encouraged by DOC to take ownership of the project. Staff at DOC have provided technical advice and support, loaned equipment, provided financial assistance and acted as advocates for the Trust's work within the local community and at local and national levels within DOC. Information is regularly shared between the Trust and DOC. The Trust has also worked with DOC staff to offer educational activities on Pomona Island though DOC's summer programme. The annual "Art in the Park" event is organised by DOC with the Trust providing evening presentations and a nature guide on the island.

DOC is represented at Trust meetings by an Advisory Trustee and Trust members are able to directly approach DOC staff. This means that advice can be obtained in a timely fashion and quickly implemented.

A second key relationship is with Tangata Whenua (Maori or iwi). Iwi have a representative on the Trust and are kept informed of progress by email, newsletters and informal discussions. Presentations are given to local iwi representatives on the Trust's restoration plans and they are consulted fully on the translocation of native species to the island.

Through its Patron, the Trust has also developed working relationships with the University of Otago. Students from the university conducted baseline research on the island prior to pest eradication and the Trust has participated in research projects and seeks technical advice from scientific experts at the university.

The Trust has encouraged its financial supporters to involve themselves in work on Pomona Island. For example, Meridian Energy, a power company, has been a major sponsor of the Trust through its Manapouri Te Anau Community Fund. By funding a "Friends of Pomona" scheme, the company has enabled the Trust to develop a fundraising strategy to ensure its on-going viability. Meridian staff have worked alongside volunteers on Pomona Island to check stoat traps and the company has also agreed to fund the transfer of mohua to Pomona Island in 2011.

The Trust has also developed a partnership with the Southland Trailer Yacht Squadron which has "adopted" the mainland trap line adjacent to Pomona. Squadron members, led by one keen individual, check the stoat traps every month, provide all the bait and sail themselves to the trap line thus reducing the Trust's transport costs. Good on-going relationships with commercial water taxi operators on Lake Manapouri enable the Trust to carry out its work on the island.

The Trust is very aware of the relationships it has with its funders and seeks, where appropriate, to keep all financial supporters fully informed of its activities. This is done through regular newsletters, the Trust's annual report and personalised emails and reports to individual funders keeping them informed about the parts of the project that they have specifically funded.

The Trust strongly emphasises community involvement. Prior to the eradication of pests from Pomona Island, views were sought from all sectors of the community about the whole project (Shaw 2006). Support came from the local council, community boards, local businesses and conservation groups. Some supporters unable or unwilling to undertake physical work on the island still show their support through the Trust's "Friends of Pomona" scheme.

COMMUNICATIONS AND FUNDING

The restoration of Pomona Island has received very good publicity with regular articles in local and regional media, extending occasionally to national media. Not all publicity, however, has been positive with the publication of a negative article relating to the Trust's planned aerial poison operation to eradicate rodents.

The website is a key means of communication for the Trust, with regular updates provided through the Trust's regular newsletter, Pomona Post. The website which receives a good level of visits is maintained and updated on a voluntary basis.

Funding for the work of the Trust has come from NZ Lottery, Transpower, Community Trust of Southland, Meridian Energy, DOC, several family trusts, two anonymous benefactors and the many Friends of Pomona. Members of the Trust are convinced that funding applications have been successful because of the charitable status of the Trust, the effort that has gone into planning the Trust's activities, the Trust's clear vision and goals, good communications and the commitment of Trustees and the local community to the restoration project.

CONCLUSIONS

The success of the Pomona Island restoration project reinforces the importance of organisation, planning, commitment, partnerships and communications. Initially, without these factors in place there was frustration at the perceived lack of progress. Once individuals were identified who had the commitment to plan and push the restoration of the island forward, progress was made. Without detailed plans at every stage, it would have taken much longer to eradicate the pests and start the re-introduction of native species. These began in 2009 with the release of 51 South Island robins (*Petroica australis*).

The partnership between the Trust and DOC has contributed significantly to the restoration of Pomona Island. DOC staff have described the Trust's achievements as "a model for community driven conservation". Such an accolade acts as a major motivator for the Trust to continue with its work.

ACKNOWLEDGEMENTS

The Pomona Island Charitable Trust would like to acknowledge the efforts of all volunteers who have worked so hard on Pomona Island. We are grateful to the Department of Conservation for their trust and support. We would also like to thank the following organisations and individuals for their financial contributions and in-kind donations: two anonymous benefactors, Meridian Energy, Community Trust of Southland, NZ Lottery, Transpower, Leslie Hutchins Conservation Foundation, Gary Chisholm Family Trust, Adventure Kayak and Cruise, Topajka Shaw Consulting, our many 'Friends of Pomona' and local family trusts.

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Developing a Regional Invasive Species Strategy for the United Kingdom's Overseas Territories in the South Atlantic

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Abstract Each of the United Kingdom's Overseas Territories (UKOTs) in the South Atlantic has a unique assemblage of endemic plants and animals, for which the greatest recognised threat is the impact of invasive species. As well as negative impacts on biodiversity values, invasive species also have significant economic impacts, particularly in those UKOTs with low annual GDP per capita. The permanent human populations of all of the South Atlantic UKOTs are small, ranging from *C.* 260 on Tristan da Cunha to *C.* 4000 on St Helena. With such low human and financial resources, it is vital to share experiences and avoid duplicating effort wherever possible. Development of a regional strategy for invasive species was seen as a key step to build links for future cooperation; especially to enable collaboration for eradication and control of invasive species in the region, and to prevent new establishment. A workshop involving representatives from all partner organisations, including representatives from agriculture, environment, and border security, along with scientists and non-governmental stakeholders, was held on Ascension Island in May 2009. This allowed a fully-consultative approach to strategy development to be taken. Priorities were developed by those attending the workshop, and consulted with other stakeholders remotely. The Regional Invasive Species Strategy will form a basis for South Atlantic invasive species work in the future.

Keywords: Strategy, invasive species, United Kingdom Overseas Territories, policy, legal framework

INTRODUCTION

The United Kingdom Overseas Territories (UKOTs) in the South Atlantic are St Helena, Ascension and Tristan da Cunha, the Falkland Islands, South Georgia and the South Sandwich Islands, and the British Antarctic Territory. St Helena, Ascension and Tristan da Cunha share a single Constitution and are legally considered to be a single UKOT; however, each has a separate Island Council, unique legislation, and unique ecology. Each of them is considered separately in this paper, and each was a separate partner in the project described below. The UKOTs have retained a connection with the United Kingdom due to the express wish of their inhabitants (FCO 1999).

In 2006, a three-year project commenced, aimed at increasing local capacity to reduce the impacts of invasive alien species on the (UKOTs) in the South Atlantic (RSPB 2006; Miller 2007, 2008; Stringer 2010). The project was funded by the European Commission's Ninth European Development Fund, and provided resources of some €2 million over the three-year implementation period. The project did not include the British Antarctic Territory due to its specific management systems and environment, but covered the other South Atlantic UKOTs. The project was led by the St Helena Government and implemented by the Royal Society for the Protection of Birds (RSPB) (Stringer 2009, 2010; Miller 2007, 2008).

The UKOTs that were included in the project are all small island states (Procter and Fleming 1999) (Fig. 1). They have small human populations of between *C.* 260 on Tristan da Cunha to *C.* 4000 on St Helena (South Georgia and the South Sandwich Islands are not permanently inhabited but there is a small transient population on South Georgia). In contrast to the United Kingdom itself, the UKOTs have a wealth of endemic species of plants and animals. Table 1 gives some background information on each of the UKOTs discussed in this paper.

Invasive alien species (IAS) have been shown to be a particularly significant threat to biodiversity on small islands such as these UKOTs (Veitch and Clout 2002; Blackburn *et al.* 2004; BirdLife International 2008), and have also been shown to have potential negative impacts on small island economies (Reaser *et al.* 2007; Jenner 2009). In 2006, there were a minimum of 2261 non-native species recorded as occurring across the UK Overseas Territories (and the Crown Dependencies of Jersey, Guernsey and the Isle of Man) (Varnham 2006). The impacts of most of these species were unrecorded. However, several avian extinctions have been recorded (Table 1), probably due to invasive species impacts (Hilton *et al.* 2001). The impacts of introduced mammals have been particularly significant (Hilton and Cuthbert 2010).

With the limited human and financial resources in the region, it was considered that, as well as taking practical action at a local level, it was vital to share experiences and avoid duplicating effort wherever possible. Development of a regional strategy for invasive species was seen as a key step to building links for future cooperation, especially to enable collaboration for eradication and control of invasive species in the region, and to prevent new species becoming established.

DEVELOPING A STRATEGY – CONTEXT

The international Convention on Biological Diversity (CBD) has identified IAS as a major cross-cutting theme. It requires Parties "as far as possible and as appropriate, (to)

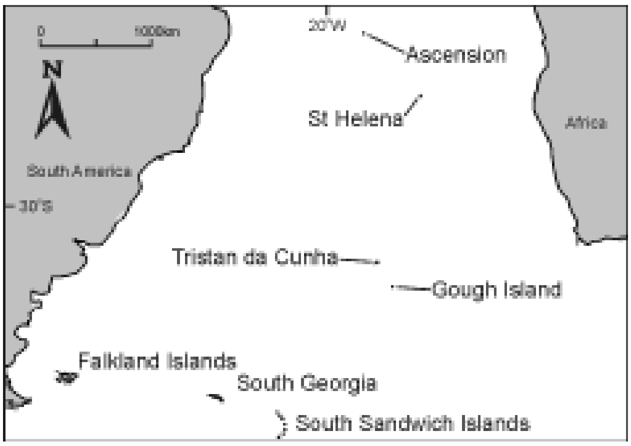

Fig. 1 The United Kingdom's Overseas Territories in the South Atlantic.

Pages 500-503 In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. Island invasives: eradication and management. IUCN, Gland, Switzerland.

Table 1 Information on the South Atlantic United Kingdom Overseas Territories (UKOTs)

UKOT	Land area (km^2)	No of islands[1]	Usual Human population	Endemic taxa[2]	Avian extinctions recorded[3]
St Helena	122	1	4000	51	8
Ascension	91	1	1000	13	2
Tristan da Cunha	201	4	260	29	2
Falkland Islands	12,173	*c.* 700	2000	32	0
South Georgia and South Sandwich Is	3903	*c.* 20	<30	3	0

[1] This number should be considered to be the "main" islands in each group.
[2] Figures from Procter and Fleming (1999), includes plants and birds, but not invertebrates as numbers are so uncertain.
[3] Figures from Hilton *et al* (2001)
St Helena, Ascension and Tristan da Cunha are considered separately

prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species" (Article 8(h)). In 2002, the CBD Conference of the Parties adopted specific Decision and Guiding Principles (Decision VI/23 on Alien Species that threaten ecosystems, habitats and species (COPVI, The Hague, April 2002)) to help Parties implement this Article. The Decision urges Parties, other governments and relevant organisations to develop IAS strategies and action plans at national and regional levels.

The UK is a Party to the CBD, and all UKOTs are included in the UK Biodiversity Action Plan 1994 which furthers CBD implementation. Individual UKOTs may take on commitments under multilateral environmental agreements (MEAs) where the UK (as sovereign state) has signed the instrument concerned and asks, at the UKOT's request, for an MEA to be extended to that territory. St Helena, Ascension and Tristan da Cunha currently implement the CBD in this way.

The main UK-UKOT framework for integrating environmental protection across sectoral policies and implementing MEAs is contained in the Environmental Charters signed by each UKOT government and the UK government on 26 September 2001. Guiding Principle 7 of each Charter is "to safeguard and restore native species, habitats and landscape features, and control or eradicate invasive species".

In response to the CBD Decision, and recognising the need for coordinated action on IAS, regional strategies have been developed by the Council of Europe (Genovesi and Shine 2004); and the Secretariat for the Pacific Regional Environment Programme (Tye 2009, Sherley 2000). The European Union (EU) has started a process that may eventually lead to publication of an EU Strategy (Brussels, 3.12.2008, COM(2008) 789 final). Strategies have also been developed by many individual countries, including by Great Britain (GB Non-native Species Secretariat 2008) and New Zealand, a country that is recognised as a world leader in its approach to invasive species (Biosecurity Council 2003). The Great Britain (GB) Strategy does not include the UKOTs in its scope; being limited to England, Scotland and Wales only (GB Non-native Species Secretariat 2008).

Table 2 Section headings in invasive species strategies (Biosecurity Council 2003; Genovesi and Shine 2004; GB Non-native Species Secretariat 2008; Tye 2009; Shine and Stringer 2010).

<table>
<tr><th>Great Britain</th><th>Pacific</th><th>European</th><th>New Zealand</th><th>South Atlantic</th></tr>
<tr><td>6. Prevention</td><td rowspan="4">C1 Biosecurity</td><td>5. Prevention</td><td>Pre-border</td><td>C Prevention</td></tr>
<tr><td rowspan="3">7. Early detection, surveillance, monitoring and rapid response</td><td rowspan="3">6. Early detection and rapid response</td><td>Borders</td><td rowspan="3">D Monitoring, early detection and rapid response</td></tr>
<tr><td>Surveillance</td></tr>
<tr><td>Incursions</td></tr>
<tr><td>8. Mitigation, control and eradication</td><td>C2 Management of established invasives</td><td>7. Mitigation of impacts</td><td>Pest management</td><td rowspan="2">E Control, management and restoration</td></tr>
<tr><td></td><td>C3 Restoration</td><td>8. Restoration</td><td></td></tr>
<tr><td rowspan="3">9. Building awareness and understanding</td><td rowspan="3">A1 Generating support</td><td rowspan="3">1. Building awareness and support</td><td>Maori</td><td rowspan="3">A Building awareness and support</td></tr>
<tr><td>Stakeholders' voice</td></tr>
<tr><td>Changing behaviours</td></tr>
<tr><td rowspan="2">10. Legislative framework</td><td rowspan="2">A3 Legislation, Policy and Protocols</td><td rowspan="2">3. Strengthening national policy, legal and institutional frameworks</td><td>Institutional arrangements</td><td rowspan="5">B Coordination, cooperation and capacity-building</td></tr>
<tr><td>Funding sources</td></tr>
<tr><td>11. Research</td><td>B3 Research on priorities</td><td rowspan="2">2. Collecting, managing and sharing information</td><td>Science</td></tr>
<tr><td></td><td>B1 Baseline and monitoring</td><td></td></tr>
<tr><td>12. Information exchange and integration</td><td>A2 Building capacity</td><td>4. Regional cooperation and responsibility</td><td>Capability gaps</td></tr>
<tr><td></td><td>B2 Prioritisation</td><td></td><td>Priorities</td><td></td></tr>
</table>

Four Strategies were analysed, and all were found to have a similar (though not identical) set of section headings, or groups of priorities (Table 2). All adhered to the hierarchical approach as recommended by the CBD Guiding Principles, and included sections on prevention, early detection, and management of established alien species. Two of the Strategies also included sections dealing with restoration, as without restoration work, sites may be reinvaded when invasive species have been removed.

All Strategies analysed gave prominence to building awareness and support, with the Bern and Pacific Strategies making this the first section in their documents. Other elements dealt with by all Strategies included legislation and institutional arrangements, research, building capacity and coordination. The need for robust prioritisation was also highlighted in two Strategies (Pacific and New Zealand).

STRATEGY DEVELOPMENT - PROCESS

The South Atlantic UKOTs represent a very small number of people (fewer than 10,000) spread over a huge area of ocean (some 40,000 square kilometres). In order to facilitate development of a South Atlantic Invasive Species Strategy, a workshop involving representatives from all partner organisations, including agriculture, environment, and border security personnel, along with scientists and non-governmental stakeholders, was held on Ascension Island in May 2009. This allowed a consultative approach to strategy development to be taken, despite a widely dispersed population. In addition to local stakeholders, a number of experts from outside the region with expertise in invasive species strategy development were invited to participate.

The key section headings / priority groupings from the Strategies that had been analysed were used as a basis for sessions during the five-day workshop. Workshop participants were asked to identify priorities in each focal area in relation to their own Territory and the region as a whole. Small "break-out" groups were used to facilitate participation from different individuals. Workshop outputs were captured electronically after each session. Participation from Tristan da Cunha was enabled by emailing session outputs to the Tristan Conservation Department daily, and feeding comments back into discussions.

Drafting of the strategy was led by Clare Shine and coordinated electronically through a web-based group established after the Ascension workshop. A draft of the strategy was submitted to South Atlantic UKOT governments in October 2009 (Shine and Stringer 2010).

CONTENT OF THE SOUTH ATLANTIC INVASIVE SPECIES STRATEGY

The South Atlantic Invasive Species Strategy largely follows the lead of the other documents discussed above. It starts by setting out an inspirational vision for the region:

"The South Atlantic is the best-kept secret in the world. Our islands, our people and our biodiversity are unique. We will work together to maintain and restore native ecosystems, prevent further damage from invasive species and to support sustainable livelihoods through actions driven by local communities, coordinated regionally and supported internationally."

This vision was drafted during the Ascension workshop, which was the first opportunity that many of the environmental professionals in the South Atlantic had had to meet. It is hoped that the networks built during this meeting will lead to future cooperative initiatives in the region.

The main sections in the Strategy are listed in Table 2, and appear in the following order:

Building awareness and support: includes actions related to securing local, UK-level and international support for invasive species work, including fund-raising.

Coordination, cooperation and capacity building: focuses on building a shared regional identity and coordinating mechanism as well as improving systems within each Territory. Establishment of a regional information exchange system and research plan are also proposed.

Prevention: includes actions related to the establishment of an effective biosecurity system for each Territory.

Monitoring, early detection and rapid response: includes actions required to develop an early warning system, improve monitoring and enable contingency planning.

Control, management and restoration: encourages the development of tools to support local management decisions, as well as including invasive species management and habitat restoration goals within government decision-making processes.

Along with the objectives, the strategy includes sections on implementation and monitoring, and general background information. Annexed to the strategy is an action and implementation plan. This includes a detailed set of tasks relating to each of the objectives, along with a lead agency or UKOT, a delivery date, and an estimate of costs where possible (Shine and Stringer 2010). Implementation will be monitored and resources will be sought externally to allow specific objectives to be achieved.

THE FUTURE AND RECOMMENDATIONS

At the time of writing, the South Atlantic Invasive Species Strategy has just been published, following final approval by Territory Councils and Governments. It is very important that the strategy is owned by local authorities, so this is a vital step. A formal launch of the strategy is now planned, and an online system for monitoring progress will be established. It is hoped that the strategy will be revised in five years time.

During the process of preparing the strategy, it was evident that there are many committed and enthusiastic people in the South Atlantic who are driving invasive species control work at the local level. However, it is also evident that the resources available in this sparsely populated region are not sufficient to deal with the enormity of some of the most pressing invasive species issues. Territory governments, non-governmental organisations and researchers from the South Atlantic and the United Kingdom should continue to collaborate to find resources for the continuation of invasive species work in the region. The Strategy provides a guide; the next phase is implementation.

ACKNOWLEDGEMENTS

The South Atlantic Invasive Species Strategy could not have been developed without the full support of all the project partners: the Governments of St Helena, Ascension Island, Tristan da Cunha, the Falklands, and South Georgia and the South Sandwich Islands, along with Falklands Conservation and the St Helena National Trust. All members of the Regional Advisory Group and enthusiastic attendees at the Ascension workshop were also vital to this process. This work was funded through the European Union's EDF-9 fund through Project No9 PTO REG 5/1; PTR 003/05/EDF IX.

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Involving the community in rodent eradication on Tristan da Cunha

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Abstract Tristan da Cunha is the world's remotest inhabited island, with a population of around 270 people. Ship rats (*Rattus rattus*) and house mice (*Mus musculus*) are present on the main island of Tristan and house mice are present on Gough Island, also part of the UK Overseas Territory of Tristan da Cunha. The impacts of invasive rodents on both islands have been well documented and detailed plans to eradicate them were developed in association with island representatives. In March 2008, the island was visited to discuss eradication plans with the island community and get their views on the proposals. Information disseminated about the project was followed by individual meetings with all government departments and other employers. These individual meetings proved by far the most effective forum for hearing people's views. Strong concerns were expressed about the safety of an aerial bait drop on Tristan, in particular the perceived risks to children, livestock and the security of the water supply. The proposed eradication of mice from Gough Island was fully supported. Although the population on Tristan did not want a full-scale rodent eradication carried out on the island, they were keen to have improved rodent control around the settlement and at agricultural sites. This work underlines the importance of detailed public consultation with small island communities during the planning of rodent eradication projects. The proposed Tristan rodent eradication project would not have been successfully completed without the full support of the Tristan community. Plans for rodent eradication on Tristan have been shelved for the time being.

Keywords: Aerial bait drop, inhabited island, house mouse, *Mus musculus*, operational plan, poison, ship rat, *Rattus rattus*

INTRODUCTION

The Tristan group is home to many endemic species, including plants, invertebrates and birds. The Tristan albatross (*Diomedea dabbenena*), now restricted to Gough Island, is one of four species of endemic birds and 27 of the islands' 50 species of native flowering plants are also endemic (Ryan 2007). Rats were introduced to Tristan in 1882 following a shipwreck and became widespread across the island within two years, while mice probably arrived sometime in the 18th century on Tristan and the 19th century on Gough (Angel and Cooper 2006 and refs therein). On Gough Island, mice prey upon chicks of the endangered Tristan albatross, Atlantic petrel (*Pterodroma inverta*) and great shearwater (*Puffinus gravis*) (Wanless *et al.* 2007), and probably also upon the chicks and eggs of the endemic Gough bunting (*Rowettia goughensi*) as well as endemic flightless moths (Angel and Cooper 2006). On Tristan the impact of rats and mice has, in general, been poorly studied. However, together with feral cats (*Felis catus*), which are now believed to be eradicated from Tristan, introduced rodents, livestock and humans are believed to be largely responsible for the historic declines in seabirds on the island (Angel and Cooper 2006). Rodents are also a pest for the human population of the island, eating potatoes as well as other crops and foodstuffs and presenting a public health risk. The continued presence of invasive rodents on Tristan also increases the risk of their reaching the nearby rat-free islands of Nightingale and Inaccessible, where they would be likely to cause further ecological devastation. If associated with conservation measures that limit human impacts on birds and the environment, the eradication of invasive rodents could thus greatly improve the security of many native species.

As the effects of introduced rodents became more obvious, Tristan's Agriculture and Natural Resources Department (ANRD) asked the Royal Society for the Protection of Birds (RSPB) to propose to the Overseas Territories Environment Programme (OTEP) a feasibility study for eradicating rats and mice from Tristan and mice from Gough.

Here we describe the results of consultation with the islanders to gauge the range of their views over rodent control and eradication options for the Tristan group. We found that the islanders supported rodent eradication, but only if there was no risk of humans or livestock coming into contact with the baits spread by helicopter. There was thus strong support for eradicating mice from Gough, but little enthusiasm for attempting rodent eradication from Tristan.

METHODS

Study site

The UK Overseas Territory of Tristan da Cunha is in the South Atlantic Ocean, approximately halfway between the tip of South America and Africa (Fig. 1). The territory consists of four islands: Tristan da Cunha (Tristan), Inaccessible and Nightingale, all within around 30km of each other, and Gough, some 350km to the south-east. The two inhabited islands, Tristan and Gough, are accessible only by ship. Tristan has been settled since the early 19th century and currently has a population of some 270, while Gough is the site of a South African meteorological station with a staff of six. The inhabitants of Tristan live on a 5 km

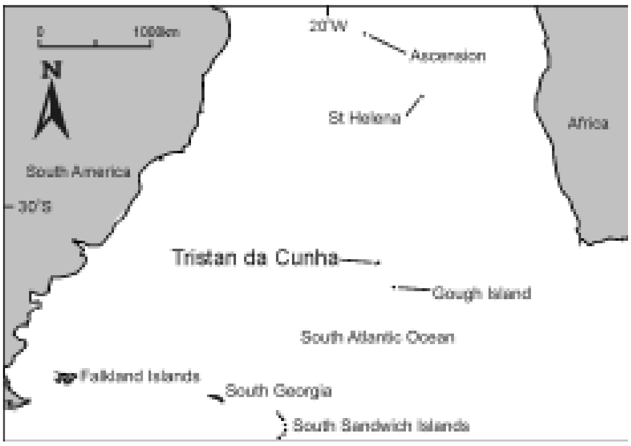

Fig.1 The location of Tristan da Cunha and Gough Islands.

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long coastal plain on the north-west of the island, where they farm cattle and sheep and grow potatoes. Additional food and other supplies are shipped to the island from Cape Town, South Africa. The islanders' main income is from the sale of fishing rights for crayfish and tourism. The island is governed by an administrator appointed by the UK government in association with an elected Island Council.

Proposed rodent eradication plans

Rodent eradication planning began in 2004 and included a stakeholder workshop in 2005 and the production of detailed operational plans for two rodent eradication projects: the eradication of ship rats and house mice on Tristan (Brown 2007; 2008) and house mice on Gough (Parkes 2007). The proposed eradication projects would be very expensive (estimated costs for the Tristan project were in excess of £2m). Funding for the eradications was to be sought only once the community had decided on their preferred options. Both plans aimed to use helicopters to spread cereal-based pellets containing the second-generation anticoagulant rodenticide brodifacoum (see Brown 2007; 2008; Parkes 2007).

Community involvement

The community on Tristan was involved at every stage of the proposed rodent eradications. The Chief Islander and another representative of the community participated in a workshop in South Africa in the early stages of the project (Anon. 2005). In addition, detailed input from every household on the island was sought by a questionnaire in 2007 (Glass *et al.* 2007) and in 2008 a consultant (KV) discussed the operational plans with the community. The discussions aimed to determine how the project could be made acceptable to the Tristan community, while avoiding any risk of failure. The discussions were approached in two ways: 1) ensuring that islanders were informed about details of the operational plans, particularly how the plans would affect people's daily lives; and 2) gathering feedback from the informed community about whether and how islanders would like to proceed with the eradication plans. There was no attempt to influence the community's decision. Rather, we wanted to make sure that people had all of the information needed to make an informed choice about the planned eradication projects. The first phase informed people of the content of the operational plans and how these projects might affect their everyday lives as well as the island as a whole. Focussing on concerns raised during the questionnaire (Glass *et al.* 2007), summaries of the projects were produced along with a list of answers to frequently asked questions, and both documents were distributed to every household. An interview about the eradication plans was also broadcast on Tristan local radio. The second phase gathered the views of island residents on the eradication plans. A public meeting, open to all residents, discussed the eradication plans and enabled islanders to comment. At this meeting, options that might make the eradication plan on Tristan more acceptable to the community were presented, based on comments from islanders and eradication planners. At the suggestion of some islanders, a series of smaller meetings were subsequently held at various workplaces on Tristan (eleven government departments and the fish factory). These meetings were collectively attended by 58 people. We did not seek to quantify the numbers of people holding particular opinions, simply to gauge the range of views of the community to the various options for rodent control and eradication.

RESULTS AND DISCUSSION

Community involvement

Awareness of the issues raised by the eradication plans varied considerably between individuals. In general, only those people connected to the ANRD had a good understanding of the aims, methods and likely ecological impacts of the eradication projects. Understandably, people tended to consider the project mainly in terms of its possible impacts on themselves, their families and their livestock. Few comments were made about the possible effects on Tristan's native wildlife.

The public meeting included proposals suggested by islanders and eradication planners such as providing water tanks for households to store water, compensation guidelines for any livestock lost due to project activities and what to do with feral stock on parts of the island. Although this public meeting was a useful way for getting large amounts of information over to the population in a short space of time, it was poorly attended and did not generate much useful feedback. However, some people were encouraged to speak to members of the Island Council or to their heads of department, which allowed their views to be passed on to some extent.

Compared with the public meeting, the smaller meetings with government departments and other employers generated much more discussion and feedback. While these meetings did not involve everyone on the island, they allowed the majority of people of working age a channel to express their views. People also had the opportunity of approaching members of the Island Council and communicating their opinions to them.

The departmental meetings revealed for the first time that many islanders had significant reservations about going ahead with plans to eradicate rodents from Tristan. Greatest concern was over the safe use of poison and this ultimately led the Island Council to decide not to take the Tristan eradication plans any further. All parties involved in the proposed rodent eradication on Tristan agreed that it could not go ahead without the support of the entire Tristan population. At the time of KV's visit, it became clear that this level of support did not exist. However, support for the eradication of mice from Gough Island was near-unanimous. Below, the main areas of concern raised by the islanders are summarised.

Questionnaire design

The household questionnaires conducted in June 2007 (Glass *et al.* 2007) showed 100% agreement in response to the question 'do you think it would be a good idea to get rid of rats and mice on Tristan', although one-third of households raised some concerns. However, during the departmental meetings in 2008 a sizeable minority of islanders stated that they were opposed to the idea of a rodent eradication project on Tristan, with several commenting that they had never thought it was a good idea. Why then had this apparently unanimous support disappeared in less than a year? We believe that while people liked the idea of Tristan being free of rodents, they did not agree with the method proposed. The questionnaire usefully identified concerns about the proposed plan, such as safety of the water supply and the risks to pets and livestock. However, the questionnaire did not specifically seek views about the way baits would be spread. Presumably everyone was told that the poison would be dropped by helicopter but only one person apparently raised any concern about 'aerial spraying' in the settlement. A direct question about whether islanders were happy with the idea of an aerial bait drop might have revealed those concerns that later emerged.

Finally, we are unsure whether peoples' perceptions were affected by their views of those conducting the questionnaire. If the community assumed ANRD staff to be in favour of rodent eradication, people may have responded more positively to their questions. In addition, the questioners' own views may have influenced the way they recorded people's responses. Such potential biases may be overcome by using professional input for designing the questions and demonstrably impartial people to carry them out. Islanders might be more comfortable speaking to people from their own community than outsiders, so staff from other local organisations could be employed to carry out such questionnaires.

Issues related to the safe use of poison

Concerns about the use of poison on the island fell into three categories: risks to island residents and livestock, methods of distribution, and persistence in the ecosystem after the eradication.

Immediate risks to residents and livestock

Many were interested to learn more about brodifacoum, its properties and its track record in eradication projects. Misunderstandings about the properties of brodifacoum were addressed, such as the widespread belief that it would poison the water supply. Evidence was presented about brodifacoum's insolubility in water and how it had never been found in samples of water taken after eradication projects (e.g., Primus *et al.* 2005). However, some fears remained including disapproval of all kinds of poison due to perceived serious, long-term and unpredictable consequences and the lack of a guarantee that previously unrecorded effects would not appear on Tristan. Some of these fears were allayed when it was pointed out that similar chemicals had been in use on Tristan for many years for pest control around the settlement with no recorded ill effects on the human population.

Another concern was that brodifacoum levels in meat and water samples could not be tested on the island. Since this process relies on the use of specialist techniques (high performance liquid chromatography, HPLC) it needs to be carried out in an accredited laboratory with the appropriate equipment. Several people raised the point that Tristan's shipping schedule meant that samples could only be tested around every 2-3 months and that there would then be at least a 7-10 day delay in obtaining results. Although unlikely, should water be contaminated, nothing could be done other than to evacuate the whole population.

Helping a community to interpret the risks of a complex project such as an island-wide rodent eradication is an extremely important but demanding task. Specialist toxicologists might have helped but a core of islanders, perhaps a majority, was wary of taking any kind of risk over the eradication. Several people commented that they wanted 100% guarantees that the project would be safe. Arguments based on the science of previous similar projects were, therefore, sometimes seen as too equivocal. Given this situation, specialists in toxicology or risk interpretation would probably not have been significantly more successful since no one could guarantee that a project would be entirely safe.

Method of distributing poison

There was also widespread concern about aerially spreading poison in the settlement, over potato patches and on pasture areas (all located on the Settlement Plain). Bait stations were perceived as a much safer option. Several islanders were of the view that, although they understood it was impractical, they would be more likely to support the project if bait stations could be used across the entire island. Islanders were also concerned that aerial bait drops had not previously been carried out on an island with such a large human population. Examples of anticoagulant bait dropped aerially on inhabited islands (Merton *et al.* 2002) in the Seychelles were considered irrelevant by most of the Tristan residents, due to the small size of their communities.

Persistence of poison in the environment

Despite information about the use of anti-coagulant poisons for around 50 years without recorded long-term health impacts, many people were unconvinced. Concerns were over persistence in the environment and long-term health consequences for humans or livestock.

Livestock

The second biggest concern was how to manage the livestock before, during and after the proposed eradication. Overall, the plans for dealing with stock on Settlement Plain (the main location of livestock on the island) appeared to be largely acceptable (i.e. building two secure areas at opposite ends of the plain and moving stock between them to avoid bait being dropped on them). Plans for reducing the numbers of feral animals (sheep on the Base and cattle at Stony Beach and the Caves) made some progress but there was no final agreement on the extent of reduction. Most people agreed with the idea of reducing stock numbers temporarily during the poisoning phase, on the condition that good-quality replacement animals would be provided.

Evacuating people from the island

Families with small children, and those with medical conditions that may leave them at higher risk from contact with anticoagulant poison, were offered the opportunity to leave Tristan for the duration of the project. They were offered places on the ship supplying the project at the start of the project and a stay in South Africa until the poisoning phase was completed. This idea seemed to be well received but several people asked how much space there would be on the project ship and what would happen if more people who met the criteria wanted to go than could be accommodated on it.

Economic threats

During the feedback-gathering phase an announcement was made by the Administrator concerning the island's economic future. Briefly, the Administrator concluded that the island was facing an uncertain economic future, with total income likely to decrease over the next few years. This statement focused people's minds on how they might have to cope with lower incomes in the future and thus become more reliant on home produced food. This made people even more sensitive to any possibility that the proposed eradication might threaten the security of traditional food sources such as island beef, mutton, fish and potatoes. If the Tristan rodent eradication project was to proceed, people would need to be assured that the plans would not threaten their economic wellbeing.

CONCLUSIONS

The proposal to eradicate rodents from Tristan was not generally perceived by the residents to have significant conservation benefits and the potential for seabird recovery on Tristan was not widely appreciated. Gough Island was seen as being much more important for wildlife, due to the presence of Tristan albatrosses and other species not present on Tristan. Although sympathetic towards Tristan's wildlife, people were more concerned about the wellbeing of their own families and livelihoods. People repeatedly mentioned that what they wanted was better pest control around the settlement and the potato patches and that this could be achieved without the risks they believed were associated with an island-wide rodent eradication project.

Such projects might be more acceptable to island communities if perceived risks could be reduced and benefits to the community increased. Perceived risks could be reduced by using bait stations around inhabited areas and by bringing livestock under cover for the duration of the project. Negotiation with islanders and education are also essential for reducing perceived risks and should be an integral part of any eradication project planned for an inhabited island. This step should include a scientific explanation of the likely ecological benefits of eradications and should focus on the ecological value of the island and its wildlife. Information about proposed eradication methods and their potential risks to humans, livestock and wildlife should also be freely available. Rodent eradication projects are essentially a package of useful people, skills and equipment, components of which could be used for the benefit of islanders. As long as there is no conflict with the needs of the eradication project, people and equipment could occasionally assist the community, something that could be built in from the planning stages. This may include providing some helicopter time for community needs in the case of projects with aerial bait drops, or shipping cargo for the community over with project equipment.

Following the many successful eradication projects of recent years, the supply of uninhabited suitable islands for rodent eradication has diminished, and increasing numbers of inhabited islands are now being considered for such projects. Dealing with communities on these islands is therefore likely to become an increasingly significant task. Every eradication project and every island community will be different, but there are common issues affecting them all. There are striking similarities between the concerns shown by the residents of different islands during the exploratory stages of rodent eradication projects (see Ogden and Gilbert 2011; Wilkinson and Priddel 2011), particularly in relation to aerial spread of poison. The methods for carrying out successful rodent eradication projects on uninhabited islands are now well defined and widely used. However, it seems that these methods will need to be modified to include avoiding poison drops over populated areas and education campaigns if they are to be successfully applied to inhabited islands.

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Rodent eradication on Lord Howe Island: challenges posed by people, livestock, and threatened endemics

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Abstract Like many oceanic islands, World Heritage listed Lord Howe Island (LHI), 760 km north-east of Sydney (Australia), has populations of invasive rodents. The house mouse (*Mus musculus*) probably arrived around 1860, and the ship rat (*Rattus rattus*) in 1918. Both species have significantly reduced the island's biodiversity. Rats are implicated in the extinction of at least 20 species (or subspecies) of birds, invertebrates and plants. Exotic rodents remain a threat to many endemic species, so much so that predation by ship rats on LHI is listed as a Key Threatening Process under New South Wales and Australian environmental legislation. A feasibility study in 2001 concluded that eradication of rats and mice was technically feasible. A cost-benefit study in 2003 demonstrated that costs of the eradication would be quickly offset by discontinuation of the current rat control programme and increased yields of commercial palm seed. A plan to eradicate exotic rodents on LHI was prepared in 2009. Technical challenges include: the presence of numerous threatened endemic species, several of which could be placed at risk during the eradication; a permanent human population of approximately 350, their pets and livestock; and a well-developed tourist industry. Several species of threatened fauna will be housed in captivity for the duration of the operation to mitigate the risk of primary and secondary poisoning. The presence of a large human settlement requires customary eradication strategies to be modified. Within uninhabited areas, bait will be aerially broadcast, whereas within the settlement, bait will be hand broadcast or placed in bait stations. Livestock will either be eliminated from the island before the eradication or aggregated into small enclosures. Community support is vital to the success of the operation, and extensive consultation is a major component of the eradication programme.

Keywords: Brodifacoum; eradication; house mouse; human inhabitants; island; mitigation; *Mus musculus*; *Rattus rattus*; ship rat; threatened endemic species.

INTRODUCTION

The Lord Howe Island Group (LHIG) is 760 km north east of Sydney, Australia. The group comprises Lord Howe Island (LHI; 1455 ha), Roach Island (15 ha), Mutton Bird Island (4.5 ha) and Blackburn Island (3 ha) plus smaller rocks and islets (Fig. 1). The first permanent settlement began on LHI in 1833. The resident population is now around 350 in approximately 150 households restricted to the central lowlands, which comprise about 15% of the island. Islanders hold perpetual leases on blocks of up to 2 ha for residential purposes, and short-term leases on larger tracts for agricultural and pastoral activities. Today, there are approximately 1000 buildings or structures on the island.

The outstanding natural beauty of the LHIG, together with its highly diverse and substantially unique flora and fauna assemblages, were recognised by its inscription as a natural World Heritage site in 1982 (DECC 2007). Tourism is one of two major sources of income, with about 16,000 visitors each year. Visitor numbers are regulated to a maximum of 400 on the island at any one time. Export of kentia palm (*Howea forsteriana*) seedlings is the other major source of income for islanders. The LHI Board (LHIB) operates a nursery that exports 2–3 million palm seedlings annually. The seed is harvested from plantations and from natural palm forests.

The first rodents to reach LHI were house mice (*Mus musculus*) in about 1860. Ship rats (*Rattus rattus*) arrived in 1918. Within two years, the rats were so widespread the Island Board of Control (a forerunner of the current LHI Board) instigated a bounty system as a means of control (Hindwood 1940). The environmental effects of the rats were immediately evident to A.R. McCulloch, who wrote that *'one can scarcely imagine a greater calamity in the bird world than this tragedy which has overtaken the avifauna of Lord Howe Island'* (McCulloch 1921).

Rats are implicated in the extinction of five species of endemic birds (Hindwood 1940), two species of plants and at least 13 species of invertebrates (Ponder 1997; LHIB

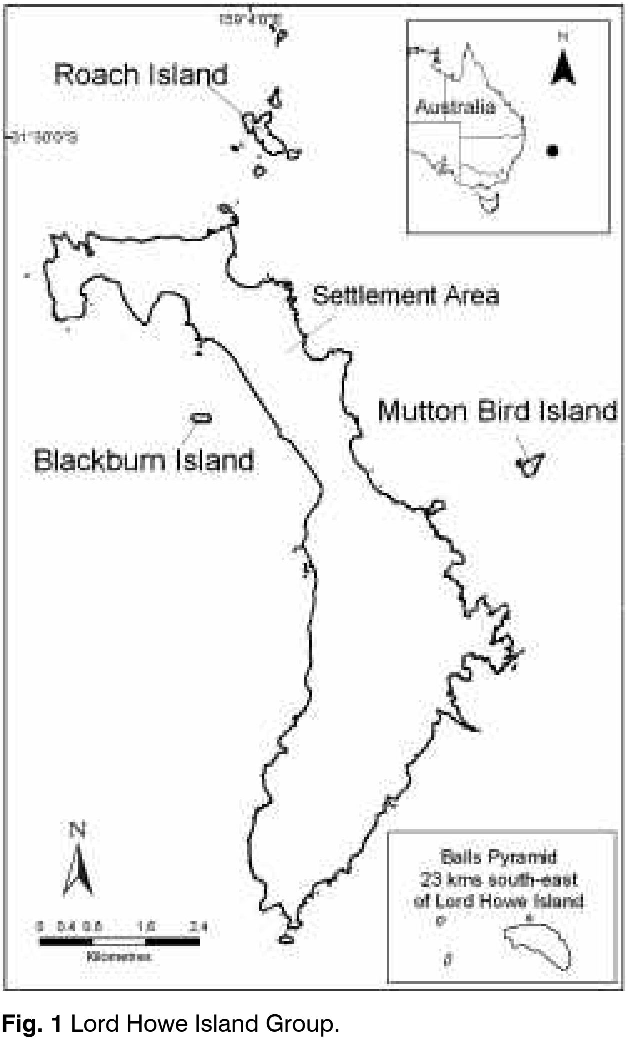

Fig. 1 Lord Howe Island Group.

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2009; C. Reid pers. comm.). Predation by rats suppresses animal populations and severely reduces recruitment of many species of plants (Moore 1966; Pickard 1982; Auld *et al.* 2010). The LHI phasmid or stick-insect (*Dryococelus australis*) disappeared from the main island; the only surviving population is on rat-free Balls Pyramid (Priddel *et al.* 2003). Likewise, the LHI wood-feeding cockroach (*Panesthia lata*), white-bellied storm-petrel (*Fregetta grallaria*) and Kermadec petrel (*Pterodroma neglecta*) are now restricted to rat-free outer islets (NSW SC 2004; DECC 2007).

The effects of house mice on the LHIG may not be as great or as well understood as those of ship rats, but are likely to be similar to those demonstrated on other islands (e.g., Newman 1994; Jones *et al.* 2003). These impacts can include direct predation on seabirds (Cuthbert and Hilton 2004), reptiles and their eggs (Towns and Broome 2003), invertebrates (Marris 2000) and seeds (Smith *et al.* 2002).

The two species of exotic rodents on LHI currently threaten at least 13 species of birds, two species of reptiles, 51 species of plants, plus 12 vegetation communities, and numerous species of threatened invertebrates (DECC 2007). Predation by ship rats on LHI is listed as a Key Threatening Process under both state and national environmental legislation. A threat abatement plan produced by the Australian Government identifies the eradication of exotic rodents from LHI as a high priority action (DEWHA 2009).

Exotic rodents also affect the social and economic wellbeing of the LHI community. The rodents host viruses, bacteria, internal parasites (such as intestinal worms) and external parasites (such as fleas, mites and lice), many of which can spread disease to humans (Henderson 2009). The island's residents continuously attempt to keep rodents out of dwellings, often through the use of poisons that pose a risk to small children and family pets requiring a level of vigilance that would be unnecessary if rodents were eradicated.

Rat predation on kentia palm seed severely reduces seed production (Pickard 1983; Billing 1999) and represents an economic loss to the island (Harden and Leary 1992). The impacts of rodents on biodiversity also have the potential to affect the island's tourism industry.

Given these effects, the LHIB embarked on eradication planning in 2006. If undertaken, LHI would be the largest, permanently inhabited island on which the eradication of ship rats and house mice has been attempted. The proposal is challenged by: 1) the complexities of targeting two pest species; 2) the existence of threatened endemic species that are susceptible to the poison; and 3) the presence of a large resident human population, a well-developed tourist industry, domestic animals and livestock.

Our paper details how the presence of threatened endemic species and human inhabitants has constrained planning and implementation of the eradication on LHI.

CONTROL OR ERADICATION?

There have been attempts to control rodents on LHI since about 1920 (Hindwood 1940). Current control is principally directed at: 1) protection of kentia palm seed over approximately 10% of the island, utilising about 1000 bait stations at 33 sites replenished five times annually with warfarin or coumatetralyl baits; 2) minimising impacts at the island's commercial palm nursery, using brodifacoum baits; and 3) reducing rodent activity in and around residences, using either warfarin or brodifacoum baits. This control effort currently costs the LHI Board around A$65,000 per annum. There have been few attempts to quantify the effectiveness of the programme, and there have been no assessments of whether there are benefits to biodiversity. Significantly, there is no control of mice beyond the settlement because this species has become resistant to warfarin (Billing 2000).

The increasing frequency and success of island eradication programmes (Towns and Broome 2003; Howald *et al.* 2007) and the increasing costs and limited success of control on LHI, led the LHIB to examine the feasibility of eradicating ship rats and house mice from the LHIG. Eradication was viewed as feasible, but the study recommended careful management of potential risks (see Saunders and Brown 2001).

In 2003, the LHIB reviewed the risks and constraints around eradicating ship rats and house mice, and to assess the various costs and benefits involved (see Parkes *et al.* 2003). This report demonstrated the financial benefits if rodents were eradicated particularly through increasing production of kentia palm seed. There were also acknowledged, but monetarily unquantified, biodiversity benefits. An eradication would thus provide overall benefits greater than can be achieved through current control programme.

A draft plan for the eradication of rodents on LHI was then developed in consultation with expert planners and practitioners from around the globe together with the LHI community (LHIB 2009). The plan recognises that: 1) eradication rather than ongoing control is the most effective long term option; 2) the impacts of rodents on the LHI environment are significant and ongoing; and 3) eradication is feasible using current techniques without unacceptable risk to non-target species and human residents.

The operation will utilise the cereal-based bait Pestoff® Rodent Bait 20R (Animal Control Products, Wanganui, New Zealand) containing brodifacoum at the concentration of 20 ppm. The primary method of bait application will be through two aerial broadcasts 10–14 days apart, with hand broadcasting or bait stations used in areas not suitable for aerial application, such as in the settlement area or where livestock are present.

MITIGATING POTENTIAL IMPACTS ON THREATENED SPECIES

Brodifacoum has been used effectively to eradicate rodents > 200 times (Howald *et al*. 2007). However, the toxin can affect some non-target species (Eason and Spurr 1995). If not mitigated, potential impacts may range from the loss of a few individuals to, on rare occasions, the loss of an entire population. Previous eradications have been accompanied by mitigation associated with the level of risk posed and the potential for population recovery (Empson and Miskelly 1999; Merton *et al.* 2002; Howald *et al.* 2005). In the latter case, any mortality associated with baiting can be far outweighed by increased survival in the absence of predation and competition from rodents. As a result, many species increase to numbers far greater than before the eradication (Empson and Miskelly 1999). On LHI, evaluation of the potential risk to non-target species, particularly to endemic species, has been a prime consideration.

Birds

Risks posed by brodifacoum to avifauna were assessed through literature reviews and non-toxic bait trials on LHI in 2007. Four endemic species of land birds survive on LHI: Lord Howe (LH) woodhen (*Gallirallus sylvestris*), LHI pied currawong (*Strepera graculina crissalis*), LHI golden whistler (*Pachycephala pectoralis contempta*) and LHI silvereye (*Zosterops lateralis tephropleurus*). The woodhen and currawong populations, which are regularly monitored, each number about 220 individuals (DECC 2007). The whistlers and silvereyes have not been surveyed, but their populations are estimated to be between 100 and 1000 pairs (DECC 2007). Without appropriate mitigation, woodhens and currawongs would be placed at risk by a baiting programme targeting rodents. These two species are both listed as vulnerable under the Australian Government's *Environment Protection* and *Biodiversity Conservation Act 1999*, and endangered and vulnerable respectively under the New South Wales *Threatened Species Conservation Act 1995*.

The woodhen is congeneric with the New Zealand weka (*G. australis*). Eradications using brodifacoum have devastated populations of weka (Brown 1997a), so woodhen are likely to be similarly affected. Blue-coloured faeces from woodhen caught during annual surveys indicate that they already consume dyed rodenticide blocks used by residents (Harden 2001). In 2007, a non-toxic bait trial conducted on LHI confirmed their attractiveness to woodhen, thus demonstrating a high probability of brodifacoum toxicosis during an eradication operation. The endemic LHI pied currawong consumes rodents, and is therefore potentially susceptible to secondary poisoning. To minimise any potential impact, at least 85% of the woodhen population, and 50% of the pied currawong population will be placed into captivity for the duration of risk. The woodhen is iconic and any avoidable loss of individuals through poisoning is unacceptable to the LHI community. A greater number of individuals will be placed into captivity than would be required based on population genetics alone.

The LHI golden whistler is at low risk given their predominantly insectivorous diet. Trials conducted in 2007 found no evidence that this species consumed baits, and secondary poisoning of a significant proportion of the population appears unlikely. The chances of secondary poisoning are further reduced by the operation being carried out in winter when invertebrate activity is low (Craddock 2003). Nonetheless, as a precaution, approximately 20 golden whistlers will be held in captivity during the eradication.

The LHI silvereye is also at low risk given their diet mainly of insects and fruit. Trials in 2007 found no evidence that this species consumed baits. Notwithstanding, like silvereyes in some New Zealand operations (Brown 1997b) a few individuals may succumb to the effects of brodifacoum. Any losses are likely to be quickly offset by increased population sizes following the release of food resources from suppression by rodents. As with whistlers, approximately 20 birds will be held in captivity during the eradication as a precaution.

The emerald ground dove (*Chalcophaps indica*) although not endemic, is less wary than the same species on the mainland, and so is considered unique. The species did not consume bait in the trial, but as a precaution, approximately 20 birds will be held in captivity during the eradication.

Based on findings from previous eradications, other native birds on LHI likely to be at risk from aerial distribution of brodifacoum baits include buff-banded rail (*Gallirallus philippensis*) and purple swamphen (*Porphyrio porphyrio*). Neither species is endemic and in the remote event that they are extirpated each is likely to recolonise. Consequently, no action will be taken to mitigate the potential effects of baiting on these species.

Birds will be held in captivity from at least one month before baiting, and until risks of primary or secondary poisoning are no longer present. The release protocol for woodhen will follow that used for weka during the Kapiti Island eradication (C. Miskelly pers. comm.). When baits have completely disintegrated (condition 6; Craddock 2004), 20 woodhen fitted with radio transmitters will be released at their site of capture and monitored for one month. If there are no problems with these birds, the remaining woodhen will be released. Helicopter support will enable rapid transfer of captured birds from the field to the captive facility, as well as their return to the wild at the completion of captivity.

Captive management will require the construction of an enclosure for woodhen and aviaries for the other species. To ensure these facilities do not provide a refuge for rodents, they will be precision built to eliminate gaps larger than 6 mm (the size required to exclude mice), and the areas surrounding the aviaries will be baited using a combination of hand broadcasting and bait stations. A trial replicating the timing and duration of the eradication will be conducted well in advance of the eradication to test the captive facilities and evaluate the methods proposed. At the completion of the trial, some woodhen will be transported to zoos on the Australian mainland. This mainland population will provide an insurance population that can be returned to LHI in the unlikely event of an unforeseen catastrophe. Woodhen have already been held in captivity both on the island and on the mainland (Miller and Mullette 1985; Lourie-Fraser 1985) and a comprehensive husbandry manual can be prepared from these experiences as well as those with weka in New Zealand. Captive management will be conducted and overseen by experienced aviculturists and veterinarians.

Reptiles and mammals

Two species of native reptiles are present on the island: LHI skink (*Oligosoma lichenigera*) and LHI gecko (*Christinus guentheri*). Both species also inhabit offshore islets around LHI and Norfolk Island, 900 km to the northeast of the LHIG. The insectivorous diet of these species (DECC 2007) exposes them to the risk of ingesting brodifacoum if they feed on invertebrates carrying brodifacoum from baits. However, the risk of secondary poisoning is low. Firstly, coagulation chemistry of reptilian blood is different to that found in mammals, and as such, the risk posed to reptiles from baiting programmes using brodifacoum is low (Merton 1987; Hoare and Hare 2006). Second, baiting will take place in winter when reptiles are less active (Craddock 2003). Third, there are no published reports of widespread deaths in reptile species following rodent eradications. In many instances the removal of rodents has resulted in substantial increases in the abundance of reptiles (Towns 1991). For example, the number of skinks on Korapuki Island increased 30-fold within five years of rats being removed (Towns 1994). Consequently, mitigation measures are not planned for reptiles on LHI.

The only extant native mammal on LHI is the large forest bat (*Vespadelus darlingtoni*) (DECC 2007), a species that is common throughout much of southern Australia (Hoye *et al.* 2008). It is insectivorous, and is therefore considered to be at low risk of poisoning.

Invertebrates

The LHIG is characterised by numerous endemic species of terrestrial invertebrates, and predation by rodents is regarded as a significant threat to many (DECC 2007). Arthropods and annelids are apparently unaffected by brodifacoum unless it is used in concentrations many orders of magnitude greater than that used in rodent eradication operations (Booth *et al.* 2001, 2003; Craddock 2003; Bowie and Ross 2006), and are not considered at risk in the LHI operation.

Although studies of molluscs indicate that they are generally unaffected by brodifacoum (Booth *et al.* 2003; Bowie and Ross 2006), one non-peer-reviewed study conducted in Mauritius reported mortality in two snail species after consuming brodifacoum baits (Gerlach and Florens 2000). Consequently, risks associated with the proposed operation were evaluated for the endemic Lord Howe flax snail (*Placostylus bivaricosus*). Results of trials indicated that *Placostylus* did not feed on bait when natural food was available. When deprived of natural food the snails consumed brodifacoum baits, but no snails died. Despite the negligible risk, *Placostylus* will be collected from locations across LHI and housed in captivity for the duration of the baiting programme. Husbandry guidelines for the care of *Placostylus* in captivity have already been established (Brescia *et al.* 2008).

In addition to *Placostylus*, four additional species of endemic land snails on LHI are critically endangered: Masters' charopid land snail (*Mystivagor mastersi*), Mount Lidgbird charopid land snail (*Pseudocharopa lidgbirdi*), Whitelegge's land snail (*Pseudocharopa whiteleggei*) and *Gudeoconcha sophiae magnifica*. Each species is so threatened by rat predation (DEWHA 2010) if rats are not removed they are likely to become extinct. The extreme rarity of these species precludes any testing of their susceptibility to brodifacoum. However, the threats to these species from not removing rodents are likely to exceed the potential risk associated with an eradication, so none of these species will be held in captivity during the operation.

EFFECTS OF HUMAN HABITATION ON ERADICATION DESIGN

A human population and their associated pets and livestock raise issues rarely encountered on other large islands where eradications have been undertaken (Towns and Broome 2003; Broome 2009). However, modifications made to ensure the safety of the community need not jeopardise the success of the operation.

Addressing livestock issues

Numbers of livestock on LHI fluctuate. Currently there are around 100 beef cattle and a herd of 14 cows provides milk for local consumption. There are also approximately 3 horses, 12 goats and 300 chickens on the island. Pigs are prohibited.

Livestock and poultry can present risks to the success of the eradication through: 1) potential human health issues associated with the consumption of contaminated beef, milk, and poultry (Fisher and Fairweather 2010); 2) stock feed, which provides an ideal harbour and food source for rodents, who may then not consume toxic bait; and 3) poultry sheds as food and shelter from which rodents may not leave. Consequently, the aim is to de-stock the island as much as possible before bait is distributed.

Livestock on LHI use approximately 75 hectares of pasture outside the settlement within which rodents must have access to bait. Australian food safety standards require that no brodifacoum is detectable in food. Consequently planning for the LHI eradication intends to eliminate the risk of brodifacoum entering the food chain.

Beef cattle on LHI will be de-stocked through slaughter during the two years leading up to the eradication. Owners will be either compensated financially or given replacement stock brought to the island when the breakdown of bait is complete. Most owners of stock have indicated their willingness to co-operate in this process.

The dairy herd will remain on the island throughout the operation, with animals confined to a small paddock connected to the existing milking shed by a narrow race. Confinement will extend until baits disintegrate. No aerial baiting using a spreader bucket will be conducted within 30 m of the holding paddocks. Instead, either aerial baiting using a trickle bucket (with a swathe width of a few metres only) or hand broadcasting will be used to distribute bait within this buffer zone. Baiting within the holding paddock will use cattle-proof bait stations. Similar arrangements will be made for goats and horses confined during the period of risk. All confined livestock will be fed with fresh cut grass from unused paddocks, alleviating the need to store food that may otherwise provide alternative food for rodents.

Brodifacoum is unlikely to contaminate milk (O'Connor *et al*. 2001). However, milk testing will be conducted after each bait drop and continue if any samples register positive for brodifacoum. Owners will be compensated for any lost milk production.

All poultry will be eliminated from the island at least one month before the eradication. Disease-free day-old chicks will be brought to the island to replace those birds removed. Although it would be more convenient to import adult chickens, quarantine measures within the LHI Act prohibits this. Poultry owners will be compensated for lost egg production.

Managing impacts on domestic dogs

There are approximately 48 domestic dogs on LHI. Cats are prohibited. Dogs are potentially vulnerable to primary and secondary poisoning. Owners will need to be vigilant to prevent animals from eating baits or consuming dead or dying rodents. To assess the risk to each dog, owners will be provided with a sample of non-toxic bait well in advance of the eradication. Any dogs that have a propensity to eat baits will need to be protected or restrained. Given the current widespread use of anticoagulant poison in the settlement area, most dog owners should be familiar with the threats posed. Nevertheless, an education programme will be implemented to advise residents of the potential risks to pet dogs and how to avoid them. The option of removing dogs from the island and housing them in boarding kennels on the mainland for the duration of risk will be available to concerned residents, at no cost. Any cases of poisoning will be treated by a course of vitamin K injections administered by the veterinarian employed for the operation.

Modifying baiting strategies to minimise risk to the community

The proposed operation on LHI will utilise a combination of aerial, hand broadcast and bait station/tray techniques in order to deal with issues associated with human habitation, public concern about aerial bating in a residential area, and to protect potable water storages. No aerial baiting will be conducted within the settlement area.

To facilitate appropriate distribution of baits around residences, the LHIB will negotiate a 'property action plan' with each leaseholder. These plans will be agreements with the LHIB about effective and safe actions on each property. These plans will detail: 1) how and where the bait will be distributed on each property (including residences, outbuildings and gardens); 2) methods to control rodents in the lead up to the eradication; 3) management of pets; 4) procedures to ensure the health and safety of all family members; and 5) procedures to dispose of compost and food waste before and during the eradication.

During the baiting period, island residents will be asked to help monitor rodent activity. Tasks include checking for evidence of bait take from bait trays and bait stations, cleaning up all rodent droppings so that any fresh droppings will be easily detected, regularly checking for signs of rodent activity, and reporting any such findings to the project team.

Managing human health issues

Brodifacoum can be harmful to humans (Fisher and Fairweather 2010) through four pathways: 1) direct ingestion of baits; 2) ingestion of contaminated food; 3) inhalation of brodifacoum-laden dust; and 4) absorption of brodifacoum through the skin. On LHI, the only pathway that poses a significant health risk is the direct ingestion of brodifacoum baits by small children. However, the low application rate (nominally 2 g of bait per m^2), the inconspicuousness of the green pellets, and the relatively large amount of bait needed to pose a serious health risk given the low concentration of brodifacoum, combine to make accidental poisoning unlikely. Furthermore, the slow-acting nature of the poison and the availability of an effective antidote, mean that baiting poses negligible risk to the community. Notwithstanding, a comprehensive human health risk assessment is currently being conducted, and will be made available to all residents.

Brodifacoum baits are already widely used within the settlement, and large quantities of warfarin bait are used at bait stations. Many of these stations are readily accessible, and currently pose an unmitigated risk to humans, particularly children. As such, residents are already familiar with the risks of consuming and handling rodenticides, and there would be little additional risk posed by the proposed eradication operation. Nonetheless, detailed information outlining the hazards associated with brodifacoum will be provided to residents before the operation. Children at the island's school will be informed about the operation and how they should behave around the toxic bait. Residents will be informed of the date of baiting well in advance, and will be issued with reminders closer to the time. Residents will also be kept informed of progress and will be notified when baits have disintegrated and there is no further risk of poisoning. A successful eradication will end the current use of rodenticides, thereby removing the risks to human health posed by the presence of rodenticides and rodents.

In the extremely unlikely event that anybody ingests bait, medical advice and aid will be provided on the island. There is a hospital on LHI and diagnostic and treatment procedures, including the provision of the antidote, vitamin K, will be discussed with the island medical doctor as part of the operational planning process.

Potential threats to tourism

Global evidence demonstrates that invasive rodents have negative impacts on native fauna and flora (Towns *et al.* 2006; DECC 2007). Such effects can diminish the natural experience offered to visitors. In some locations, the impact of invasive rodents on tourism has provided the impetus for rodent eradication. For example, in the Seychelles Islands, which are a global biodiversity hotspot, the importance of rat eradication to tourism is well recognised, and resort owners acknowledge that 'exclusive five-star tourism and rats don't mix' (Nevill 2004).

Since tourism is the primary revenue earner on LHI, and the island's unique biodiversity underpins its World Heritage status, one might expect that improving experiences with biodiversity would be extremely important to the community. Surprisingly, some tourism operators view rodents as having little or no impact on biodiversity. Furthermore, there is some concern that publicly announcing the intent to eradicate rodents will irrevocably damage business opportunities. This view contrasts with experiences in the Seychelles, where tour operators embraced eradications as a means of enhancing their tourism experience (Nevill 2004). Further engagement with the tourism industry is needed to explore potential opportunities and ensure that there is no downturn in tourism arising from the eradication operation on LHI.

Transport to and from the Island and its implications for biosecurity

Natural reinvasion of LHI by rodents is impossible due to the island's approximately 500 km distance from the Australian mainland. However, the island is serviced by fortnightly cargo ships from the mainland, as well as daily commercial freight and passenger flights. There are also irregular visits from yachts and private or military aircraft. Commercial schedules, combined with a requirement of visiting boats and aircraft to notify the local authorities of their proposed arrival, ensures that the timing and potential source of invasive species arriving on the island are known.

A biosecurity strategy (Landos 2003) currently operates on LHI. Additional measures needed to ensure that rodents are not reintroduced once they have been eradicated include: 1) improved checks of cargo before departure from the mainland; 2) in-transit checks of sea freight; 3) pre-landing inspections of the cargo vessel and private yachts; and 4) arrival inspections of all aircraft and passengers using trained detector dogs. These measures are to be introduced before the eradication begins, but should also help prevent other unwelcome flora and fauna from reaching the island. The introduction of exotic pests has been identified as an ongoing threat to the biodiversity of the LHIG and prevention is a high priority (DECC 2007).

Some community members are concerned that increased biosecurity measures would impose additional inconvenience on visitors and residents, and increase the already high cost of living. On the other hand, the social and environmental costs of invasive species can be immense, as is the cost of controlling or eradicating them. Community education will further emphasise the importance of enhanced biosecurity to protect the environment, and links with LHI's World Heritage status and tourism industry.

Socio-political issues and eradication planning

Support for a rodent eradication from residents of an inhabited island is most likely if the threats posed by rodents are understood, the eradication seems possible, and the benefits that will accrue are appreciated. Support is likely to be strongest if the eradication will demonstrably provide benefits to the island's biodiversity and its inhabitants.

Several community meetings and focus groups have been held on LHI to inform the community about the need for an eradication, how it would be undertaken and when it was likely. The meetings outlined environmental benefits of rodent eradication, along with the potential flow-on effects for tourism. We explained that planning for the operation utilised best-practice procedures and drew on a wealth of previous experience gained in successfully eradicating rodents from islands. We identified the potential risks to the community and to the environment, and outlined the contingencies built into the planning process to ensure that these risks were mitigated. We also explained the ongoing risks to children, non-target species, livestock and pets associated with the continued use of rodenticides should the proposed eradication not be undertaken.

A survey on LHI in mid 2009 , approximately 15 months after the commencement of consultation, indicated that a minority of residents believed exotic rodents to be either a benign addition to LHI, or in some kind of "equilibrium" with other species. However, most people (96% of 126 respondents) agreed with the need to eradicate rodents from the island, although understandably some questioned its feasibility. Most residents were generally supportive of the methods proposed, although many expressed concerns, particularly in relation to public safety. The fact that LHI will be the largest permanently inhabited island on which a rodent eradication has been undertaken has led some to believe that the operation is an experiment in which they are "guinea pigs".

The issue of incidental non-target mortality highlighted differences between the values of resource managers and those held by some members of the community. Planning includes mitigation measures for those species where a population level risk is likely and the species is of conservation concern. In the case of susceptible introduced species, such as blackbirds (*Turdus merula*), no mitigation is planned. Some residents view the death of any birds by baiting as unacceptable, making no distinction between endemic, native, and introduced species, nor acknowledging the current predation of LHI birds by rats and mice. Conflicting value judgements by resource managers and local communities are not uncommon (Parkes *et al.* 2002; Howald *et al.* 2005).

A few respondents to the 2009 survey suggested that the current control programme should be either continued or expanded, apparently failing to appreciate the difference between control and eradication. This is not surprising given that natural resource managers sometimes also fail to comprehend the difference (Thomas and Taylor 2002). Notwithstanding, because rodent eradication is achievable on islands, it seems illogical to elect for ongoing control that has little biodiversity benefit, which would perpetually place toxins in the environment, and to which rodents are developing immunity.

Many concerns raised by the community can be addressed through appropriate information. Fact sheets dealing with different aspects of the eradication have now been produced and distributed. Topics include: 1) impacts of rodents on islands; 2) the benefits of rodent removal; 3) the impacts of baiting on non-target species; 4) the choice of poison; 5) the methods of bait dispersal; 6) human health risks; and 7) risks to the marine environment. Some concerns from the community have required amendments to the original eradication plan. The challenge is to incorporate such modifications without jeopardising the success of the operation.

Freely available, detailed, and summarised information should in theory allay most concerns within the community. Unfortunately, incorrect information distributed by a few vocal detractors has created confusion and engendered some unjustified fear in the community. The detractors even alleged corrupt activities, which after investigation by Australian authorities were dismissed as baseless. The incident does highlight the extent to which some residents will attempt to discredit the planned operation.

In summary, there is ample evidence that the eradication of exotic rodents on LHI is achievable and potential threats to non-target endemic species can be overcome. The biggest remaining challenge involves reversing misconceptions and fully engaging the local community. If this can be achieved, the removal of exotic rodents from this World Heritage site will be arguably one of the most significant management actions undertaken for threatened species conservation in Australia.

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CLOSING ADDRESS

2001 to 2010 and beyond: Trends and future directions in the eradication of invasive species on islands

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Abstract My interest in island conservation grew out of work to conserve Australia's mammals; the shocking figure of 22 extinctions since 1788 would have been 30 without populations on continental islands. My efforts to eradicate invasive animals on islands commenced in 1969 with the eradication of rabbits (*Oryctolagus cuniculus*) on the 19 ha Carnac Island. Eradication of goats (*Capra hircus*) on the 4267 ha Bernier Island and ship (black) rats (*Rattus rattus*) on the 24 ha Bedout Island were followed by many others in Western Australia. The last one I led was the eradication of rats and feral cats (*Felis catus*) on an archipelago – the Montebello Islands. Recently, I have participated in developing a quarantine management system for the 23,000 ha Barrow Island. This personal journey mirrors, to some extent, the development of island management for biodiversity conservation worldwide. Island management for biodiversity conservation is very important. Islands contain a disproportionate share of the world's terrestrial species, including many endemics; islands are vital breeding places for seabirds, sea turtles and seals; islands are especially vulnerable to the impact of invasive species; eradication of invasive species is possible on islands; and successful conservation actions, especially eradication of invasives, are among those with greatest benefit to biodiversity at the least cost. A comparison between papers delivered at the 2001 and 2010 conferences shows that more nations are conducting invasive species eradications; a wider array of invasive species is being addressed; larger and more remote islands are now the subject of invasive species work and more projects are being conducted on inhabited islands. Future issues that remain unresolved include rat eradication on tropical rain-forested islands; dealing with difficult species such as tramp ants and mongoose; reducing impacts on non-target species; dealing with inhabited islands, animal welfare and ethics; properly documenting costs and benefits; and the implications of project failure. Prevention is better than cure and island biosecurity is becoming increasingly important. The 2010 conference has demonstrated the importance of managing invasive species on islands across the spectrum of prevention (biosecurity/quarantine), detection, control and eradication, plus the necessary post-project monitoring. It is clear that managing biodiversity on islands is extremely cost effective and it is not surprising that interest in this subject is increasing worldwide. Simberloff's challenge in the keynote address at the 2001 conference: '*Today Tiritiri Matangi, tomorrow the world!*' is as relevant today as it was then.

Keywords: Biosecurity, invasive species, rabbits, *Oryctolagus cuniculus,* goats, *Capra hircus,* ship rats, *Rattus rattus*, feral cats, *Felis catus*

INTRODUCTION

I was privileged to work for a series of Western Australian conservation agencies for more than 30 years as a research scientist and manager, but never during that time did my job description include the word 'island'. So, how did I become involved in island conservation and the eradication of invasive species on islands?

My interest in island conservation started with work to conserve Australia's mammals. Since European settlement of Australia began in 1788, 22 species (7%) of terrestrial mammals have become extinct. Without islands, however, this already-shocking figure would be 30, as eight species that became extinct on the mainland persisted on continental islands (Burbidge *et al*. 2008). Australian islands also have secure populations of many indigenous mammal species that are threatened with extinction on the mainland (Burbidge 1999). These extinctions and declines are, to a great extent, due to invasive species, primarily predators, such as the European red fox (*Vulpes vulpes*) and feral cats (*Felis catus*), but also herbivores, such as rabbits (*Oryctolagus cuniculus*), sheep (*Ovis aries*), goats (*Capra hircus*), pigs (*Sus scrofa*) and cattle (*Bos taurus*) (Burbidge and Manly 1999; McKenzie *et al*. 2007). The rate of extinctions in Australia suggests that, from an ecological point of view, it is the world's largest island, not a continent. The role of introduced rats and mice in causing extinctions on 'continental' Australia is uncertain.

I would like to start by briefly outlining my personal journey in island management, as this, to some extent, mirrors the development of invasive species management on islands worldwide. My first involvement with the eradication of an invasive species on an island was in 1969. European rabbits were destroying vegetation on Carnac Island (19 ha), a nature reserve in the Indian Ocean near Perth. Carnac Island is important for seabird nesting, and the nesting burrows of one species, little penguin (*Eudyptula minor*), were collapsing. After consulting with vertebrate pest researchers in the State's Agriculture Protection Board, we introduced the myxoma virus to the island's rabbits, in the hope that they had not previously been exposed to it and had no immunity. Immunity was present, so the next attempt was to use a toxin; in this case 1080 in carrots. After a couple of days of feeding with toxin-free carrots in late summer (when food was limiting), a single feed of carrots with 1080 was effective.

My next job was very much concerned with mammal conservation. Bernier and Dorre Islands in Shark Bay have populations of five endangered mammal species, most of which are extinct on the mainland. Around 1900, before the islands were included in the protected area system, goats (*Capra hircus*) were introduced to both islands; however, they persisted only on one – Bernier (4267 ha). There they were competing with native mammals for food, destroying the mammals' diurnal shelter and causing erosion. In 1969, we commenced shooting the goats and in subsequent years intensified this option. However, despite using fixed-wing aircraft to muster goats towards shooters, and despite assistance from a platoon of Ghurkhas of the British Army,

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ground-based shooting, while adequate for control, clearly was not going to achieve eradication. By the early 1980s, helicopter shooting of feral donkeys (*Equus asinus*) and feral cattle was underway in northern Australia and we were able to employ an experienced pilot-shooter team on Bernier Island. They succeeded in eradicating the remaining goats in three days.

Then I moved to rats. Ship rats (*Rattus rattus*) had been introduced to many islands in the north west of Western Australia, presumably from the many small pearling vessels that were active in the latter part of the 19th Century. Bedout Island (24 ha) was one such island and is important for seabird breeding. While the breeding success of larger seabirds such as brown and masked boobies (*Sula leucogaster* and *S. dactylatra*) and lesser frigate-birds (*Fregata ariel*) was unaffected, smaller species such as common noddy (*Anous stolidus*), and roseate and sooty terns (*Sterna dougallii* and *S. fuscata*) had abandoned the island. In 1981, after again taking advice from local vertebrate pest experts and examining the literature on island eradications, we used oats, vacuum-impregnated with Pindone as the bait and laid it on a grid over the island. Follow-up surveys confirmed eradication and since then there have been reports of recolonisation by common noddy and sooty tern.

Many other island eradications followed, including rats, mice, foxes, rabbits and cats. These are summarised in a paper presented at the 2001 conference (Burbidge and Morris 2002). The last one that I led was an eradication of ship rats and feral cats in the Montebello Islands. Every island eradication has its unique issues—the Montebellos certainly did. In 1952 and again in 1956, the islands were used by the British for the testing of three nuclear weapons; while it is now safe to visit, safe working procedures are necessary near the three ground zeros where residual radiation persists. The remoteness of the Montebellos and their convoluted shape added to planning difficulties. The operation commenced in 1996. Ship rats were treated first and because of the presence of two granivorous birds that may take rodenticide, we opted for ground baiting with a commercial rodenticide with brodifacoum as the active ingredient. Some of the remoteness issues were solved by local oil and gas companies helping with logistics, especially transporting gear in barges. The 100 or so islands totalled over 2100 ha and, with a 50 m grid, required >11,000 bait stations. A helicopter was used to place equipment dumps on all the larger islands and to access some of the islands more remote from our base; then a small boat was used to access the remaining islands. Volunteers, mostly from the agency's staff who gave up some of their holidays, were crucial. In all, more than 40 people took part, and the operation lasted four months.

Two years later, monitoring revealed that rats remained on the largest island, Hermite (1020 ha) and on two small adjacent islands. We were able to rebait the smaller islands, but Hermite was beyond our capacity. Non-target issues had proved negligible, allowing a switch to a helicopter-borne spreader bucket. Advice and assistance from the New Zealand Department of Conservation made our planning and execution much easier. After some initial problems with gear and navigation, helicopter baiting in 2001 completed the eradication of ship rats. Feral cat eradication was achieved in 1999 via a combination of aerial baiting and trapping (Algar *et al*. 2002), so the archipelago is now free of invasive animals.

Two highly threatened native mammals, mala *Lagorchestes hirsutus* and djoongari *Pseudomys fieldi*, were introduced to Trimouille and North West Islands respectively in 1998 and 1999 (Langford and Burbidge 2001), and in 2010 spectacled hare-wallabies (*Lagorchestes conspicillatus*) and golden bandicoots (*Isoodon auratus*) were reintroduced to Hermite Island from nearby Barrow Island. Two birds, spinifexbird *Eremiornis carteri* and Barrow Island black-and-white fairy-wren *Malurus leucopterus leucopterus*, are also being translocated to Hermite.

Barrow Island (23,000 ha), off the north west of Western Australia, is one of the most important island conservation reserves anywhere in the world, with several threatened mammal species, many endemic taxa, sea turtle rookeries and unique ecosystems. It has had a producing oilfield on it since the 1960s and a quarantine system to protect its values developed during the early years of development. After the eradication of ship rats from a small portion of the island (Morris 2001) it is now one of the largest land masses in the world with no introduced mammals. Recently, the Western Australian and Australian governments approved the construction of a large liquefied natural gas plant on the island. The greatest risk of such a large development (estimated cost AU$43billion, up to 3500 construction workers on the island) is the introduction of invasive species, and one condition of approval was the development and approval of a quarantine management system (QMS). Chevron Australia, the gas plant operator, committed to prepare a 'beyond world's best practice' system, based on a risk management and pathway analysis approach. With the aid of consultants and two Quarantine Expert Panels, they identified and analysed 15 pathways by which people, equipment and food would arrive at the island and its surrounding waters (Stocklosa 2004). I was a member of both expert panels and attended many of the expert workshops that analysed pathways and advised on barriers. I am currently a member of a new Quarantine Expert Panel set up by government to advise on the completeness and implementation of the QMS. While I was aware of the need for high-quality island biosecurity to prevent invasive species arrival or reinvasion after eradication, this project has heightened my awareness of the multiple ways that invasive species can arrive, and how to prevent their arrival.

WHY ISLANDS?

We should remind ourselves why we are attending this conference: why the conservation of biodiversity on islands is so important. Eradication of invasive species on islands is not being undertaken so we can break records for the largest island or for the most species eradicated; it is a means to achieve biodiversity conservation. Managing islands is important because:

- Islands contain a disproportionate share of the world's terrestrial species and have many endemics (Myers *et al*. 2000; Johnson and Sattersfield 2008; Genovesi 2011).
- Marine animals, such as seabirds, sea turtles and seals, need land to reproduce and many breed only, or substantially, on islands.
- Islands are especially vulnerable to the impact of invasive species. Most extinctions have been on islands and invasives are the major cause: 'The majority of

Table 1 Number of papers (including poster papers in 2010) that deal with eradication and control of invasive species on islands by nation.

Nation	2001	2010
New Zealand	28	33
United States of America	18	30
Australia	9	12
Mauritius	6	0
United Kingdom	2	3
Ecuador	2	3
Mexico	2	3
Japan	1	13
Seychelles	1	0
France	1	5
Puerto Rica	1	0
Nauru	1	0
Canada	0	2
Fiji	0	2
Sri Lanka	0	2
Brazil	0	1
Chile & Argentina	0	1
Kiribati	0	1
Samoa	0	1
Yemen	0	1

Note: Overseas territories, including self-governing territories, of the UK, USA and France included in those countries.

recorded species extinctions since 1500 AD have occurred on islands. A total of 72% of recorded extinctions in five animal groups (mammals, birds, amphibians, reptiles, and molluscs) was of island species. Furthermore, for each individual taxonomic group the percentage of recorded extinctions occurring on islands was greater than that occurring on continents. In total, 62% of mammals, 88% of birds, 54% of amphibians, 86% of reptiles, and 68% of molluscs were island species.' (Baillie *et al.* 2004).

- Unlike continental land masses, eradication of invasive species is, with good planning and execution, possible on islands and the potential for reinvasion is low if good quarantine is in place (Clout and Veitch 2002).
- Island restoration may be possible (e.g., Towns and Ballantine 1993).
- Successful conservation actions on islands are among those with greatest benefit to biodiversity at the least cost (Genovesi 2011).

Table 2 Invasive species targeted by phylogenetic group in papers in the 2001 and 2010 conferences.

Group	2001	2010
Mammals	42	84
Birds	0	1
Reptiles	1	4
Amphibians	1	3
Fish	0	2
Insects	6	3
Molluscs	1	0
Plants	13	5

A DECADE OF PROGRESS: COMPARING THE CONFERENCES

The 2001 Conference that resulted in the book *Turning the Tide: The Eradication of Invasive Species* (Veitch and Clout 2002) was a landmark in the management of islands for biodiversity conservation. It has become necessary reading for practitioners, especially in relation to invasive vertebrates. Importantly, the book and its papers are freely available on-line. Comparing the papers and posters presented at the 2001 and 2010 conferences should reflect the progress in island management over the past decade.

The number of papers and posters presented has increased significantly, from 72 in 2001 to 138 in 2010. Examining the number of papers that deal with the eradication or control of invasive species by nation shows that the number of different nations represented at the conferences has also increased, from 12 at the 2001 conference to 21 in 2010 (Table 1). Additionally, in 2010, one paper reviewed work in Europe, three papers dealt with multiple Pacific Islands, and several other papers covered specific issues rather than concentrating on particular islands. At both conferences, papers about New Zealand islands predominated (28 in 2001, 33 in 2010), partly because New Zealand has led the world in island eradications, but also partly because both conferences have been held in New Zealand. There are other noteworthy differences. For example, the number of papers dealing with islands within the United States of America increased from 18 to 30, and the number of papers

Table 3 Mammal species targeted for eradication and control in papers in the 2001 and 2010 conferences.

Mammal species	2001	2010
Arctic fox *Alopex lagopus*	X	
Arctic ground squirrel *Spermophilus parryii*	X	X
Beaver *Castor canadensis*		X
Crab-eating macaque *Macaca fascicularis*	X	
Elk *Cervus canadensis*		X
European rabbit *Oryctolagus cuniculus*	X	X
Feral cat *Felis catus*	X	X
Ferret *Mustela putorius furo*		X
Gambian giant pouched rat *Cricetomys gambianus*		X
Goat *Capra hircus*	X	X
Hedgehog *Erinaceus europaeus*		X
Small Indian mongoose *Herpestes javanicus*	X	X
Marmot *Marmota* sp.		X
Mink *Neovison vison*		X
Musk shrew *Suncus maurinus*	X	
Muskrat *Ondatra zibethicus*		X
Nutria *Myocastor coypus*		X
Pig *Sus scrofa*	X	X
Possum *Trichosurus vulpecula*	X	X
Raccoon *Procyon lotor*		X
Rats and mice – *Rattus rattus*, *R. norvegicus*, *R. exulans*, *Mus musculus*	X	X
Red deer *Cervus elaphas*		X
Red fox *Vulpes vulpes*	X	X
Red-bellied squirrel *Sciurus aureogaster*		X
Reindeer/caribou *Rangifer tarandus*	X	X
Rock-wallaby *Petrogale penicillata*	X	
Sheep *Ovis aries*	X	X
Stoat *Mustela erminea*		X

dealing with Japanese islands increased from one to 13. On the other hand, the number of papers dealing with invasive plants decreased from 13 to five.

The number of invasive species targeted has also increased (Table 2). It is notable that most papers from both conferences deal with invasive mammals, especially murid rodents (rats and mice). A breakdown of the mammal species targeted (Table 3) shows that a greater number of species were the subject of papers in 2010 than in 2001. The range of species covered in papers from both conferences reflects the propensity of humans to move mammals around the world, both purposefully and accidentally.

Papers covering the eradication or control of single and multiple invasive species on islands changed only a little in proportion between the two conferences: in 2001 there were 36 single species papers and 32 multiple (47%) species papers, while in 2010 there were 73 single and 45 multiple (38%) papers.

The 2010 conference was also notable for the number of papers on social and economic issues, with 17 papers compared with one in 2001, a change that is particularly important with the trend towards dealing with invasive species on inhabited islands. Also, a notable feature of 2010 was the application of new technologies to invasion tracking e.g., DNA fingerprinting and mathematical modelling for detection theory.

So, what trends can be deduced from the above statistics and a reading of the papers?

- More nations are conducting invasive species eradication and control on islands.
- A wider array of invasive species is now being addressed.
- Islands with single and multiple invasive species are still being treated.
- Larger and more remote islands are now the subject of invasive species work.
- Eradication projects are being planned or conducted on more inhabited islands.
- The increasing importance of considering social and economic issues when planning island eradications.

FUTURE ISSUES

From papers presented at the conference it is clear that many issues remain unresolved. Some of these are:

- Eradication of rats on tropical rain-forested islands. The presence of land crabs on such islands presents major difficulties as they consume standard rodenticide baits. Rats may also reside in trees and not be able to access bait laid on the ground. Development of a crab deterrent to add to bait would be a major step forward (Wegmann *et al*. 2011).
- Some species are particularly difficult to eradicate, e.g., tramp ants (Boland and Smith 2011; Inoui *et al.* 2011; Randall and Morrison 2011) and the small Indian mongoose, particularly on larger islands (Peters *et al.* 2011; K. Ishida pers. comm.; S. Sasaki pers. comm.; F. Yamada pers. comm.).
- Non-targets remain a major issue for many islands. While many novel techniques have been developed, especially bait stations designed to prevent access by non-targets while allowing access to bait by the invasive species, these may not work where the non-target species is smaller than the target, or can climb as easily. The establishment of 'insurance' colonies is one way of surmounting this issue. The development of baits with deterrent additives for non-targets, such as birds, would be a major step forward.
- Islands with resident human populations are a special case, as the use of toxins may present real or perceived human health risks, or risks to domestic or companion animals. Residents must be involved in planning eradications and those proposing to conduct the eradication need to present unbiased evidence about risks and benefits. If the proponent is a government organisation, it may be important for environmental non-government organisations to become involved to counter-act the frequent mistrust of government.
- Animal welfare and ethics issues are becoming more important and the discussion of this issue by Cowan and Warburton (2011) is timely. Animal 'rights' activists have the potential to disrupt eradication projects or, by using the news media, pressure political leaders to cancel projects. Counteracting these emotional arguments is possible only by ensuring that the public and news media are fully aware of the benefits of carrying out the project and the costs of not doing so.
- Allied to this is the need to properly document the benefits that island eradications bring to conservation. These have not always been clearly measured or publicised (Lorvelec and Pascal 2005; Towns 2011).
- The implications of eradication project failures are becoming more important. Proposals to conduct invasive species eradications on more and larger islands will cost larger amounts of money, usually public money, and failures will strengthen arguments not to spend funds on eradications. Practitioners need to be particularly careful to assess the risk of failure and not to proceed if the risk is too high.

BIOSECURITY

Prevention is better than cure. This axiom applies to island management, as it is better to prevent invasive species arriving on islands than to have to eradicate or control them after arrival, with consequential major costs, which can be financial and sometimes environmental. Increasing world trade and lowered trade barriers plus increasing human mobility mean that the risk of non-indigenous species arriving accidentally on islands is increasing. Having high-quality and effective quarantine management systems in place for high-value islands is thus increasingly important. Biosecurity is also a vital component of island eradication plans as successful eradication can be negated by reinvasion.

- Biosecurity should be in place before eradication occurs (e.g., Simberloff 2001).
- Biosecurity is necessary for inter-island trade, e.g., Guam, Kiribati.
- Biosecurity programs should be a requirement of approval of developments on islands, e.g., Chevron Australia's Gorgon Gas Project on Barrow Island, Australia (Stocklosa 2004), and The United States Department of Defense plans for Guam and Micronesia (Feidler and Andreozzi, 2010 conference side meeting).
- Public education, especially of boat owners, is an essential tool to limit further invasions (Broome 2007).

The increasing importance of island biosecurity (as opposed to national biosecurity which is usually designed to protect primary industry rather than biodiversity) indicates the need for more interaction between practitioners. Perhaps this should be a feature of the next Island Invasives conference?

CONCLUDING REMARKS

This second world conference on Island Invasives initiated by the Invasive Species Specialist Group of the IUCN has demonstrated the importance of managing invasive species on islands across the spectrum of prevention (biosecurity/quarantine), detection, control and eradication, plus the necessary post-project monitoring. It is clear that managing biodiversity on islands is extremely cost effective and it is not surprising that interest in this subject is increasing worldwide. Simberloff's challenge in the keynote address at the 2001 conference: '*Today Tiritiri Matangi, tomorrow the world!*' is as relevant today as it was then.

It is also clear that projects aimed at eradicating invasive species on islands will become more common, but also more complex and expensive as larger, more remote islands with more than one invasive species are tackled. A continuing need for cooperation and coordination between eradication and control experts is indicated. Learning from each other is the way forward. I would like to thank the conference organisers for helping this happen.

ACKNOWLEDGEMENTS

My thanks go to the 2010 conference organising committee for inviting me to present the closing address and to Landcare Research for travel and accommodation support. I am indebted to the many colleagues and volunteers who have made the eradications of invasive species and other projects mentioned in this paper a possibility and a success.

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Abstracts

These abstracts are for papers which were presented at the conference, either as oral presentations or poster papers, but for which the authors have chosen not to prepare and publish a full written paper.

These abstracts are given in the alphabetical order of the prime author of the paper with the address of only that first author included.

Improvement of a kill trap for mongoose eradication projects on two islands in Japan

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The small Indian mongoose (*Herpestes javanicus*) was established on Okinawa Island (1206km^2) in 1910 and on Amami-oshima Island (712 km^2) in 1979. In 2000, national and prefectural governments launched a mongoose control project on both islands. In 2005, the Invasive Alien Species Act was enforced in Japan and a ten year eradication programme launched. By 2009, this eradication project was in its fifth year. Adequate trapping is important but live trapping techniques are too labour intensive to use over large areas. We began using kill traps in 2003 on Amami and 2008 on Okinawa and gradually increased their numbers. However, a species of endemic bird and two species of rat were captured as non-target species so the traps have had to be repeatedly improved. The two native rat species, which inhabit Amami and Okinawa, are also affected by mongoose introduction and their distribution range is reducing in the areas where mongoose are abundant. Remodeled kill-traps enable us to avoid unintentionally catching native birds, but it is difficult to avoid catching the rats. Therefore kill-traps and live traps were used separately depending on the areas and the seasons when rats were active. Now that mongoose density has decreased to low levels, some native animals including rats are recovering. While the native rats recover, the trapping area where we can use kill-traps is declining. We now need additional improvements to trap design, or good lures for the mongoose, in order to avoid detrimental effects on native rats.

Potential operational evolution in pest eradication through use of a self-resetting trap

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Eradication and management of stoats *(Mustela erminea)* and rats *(Rattus rattus* and *R. norvegicus)* is of vital importance to biosecurity in New Zealand. Kill trap operations have proved the ability to eradicate and control populations sufficient for the protection of native species but require intensive and continued maintenance and expense. Goodnature Limited and the Department of Conservation collaborated to develop a self resetting trap for stoats and rats to exceed the annual performance of current trap schedules with no human intervention, be lightweight, durable and user friendly. Development and testing was completed in June 2009 resulting in a new control tool which kills, clears and resets twelve times before requiring human intervention. This development allows entire control networks to achieve a 'knockdown period' and then remain 100% available to pest predators, dramatically reducing labour required in operation set up and maintenance. It is speculated that this tool will lead to new operational strategies allowing eradication and management of rats and stoats in significantly larger areas.

Multi-threat control strategies for endangered species management on O'ahu army lands in Hawai'i

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The U.S. Army Garrison Hawai'i is required to manage 67 endangered taxa, including 51 plants, nine tree snails, one bird species, and potentially six picturewing flies on the island of O'ahu, Hawai'i. These species occupy fragmented, disturbed habitat and face multiple threats. The O'ahu Army Natural Resources Program (OANRP) manages these species across 56 geographically defined Management Units (MUs). Located on the rim of Makua Valley, the Kahanahāiki MU encompasses 36.4 ha (90 acres) of mixed native/invasive mesic forest and is home to one tree snail species and both wild and reintroduced populations of 10 endangered plant taxa, including *Cyanea superba* ssp. *superba*, which was extirpated from the wild in 2003. Threats include feral pigs (*Sus scrofa*), ship rats (*Rattus rattus*) and Pacific rats (*R. exulans*), mice (*Mus musculus*), weeds, snails, slugs, and arthropods. The goal of threat control is to restore habitat in the MU such that endangered taxa thrive and maintain viable, stable populations. Multiple threats must be controlled simultaneously to achieve this goal. Feral ungulates were successfully excluded from the area in 1997 via fencing and snaring. A large snap trap grid, installed in early 2009, maintains low numbers of rodents. Weeds are primarily managed around rare taxa, although more aggressive restoration projects seek to create more continuous native forest. Both incipient and established weeds are controlled. Invasive slugs, predators of native seedlings, are controlled using a natural product containing iron phosphate. Native tree snails are protected from the carnivorous snail *Euglandina rosea* via multiple barrier (salt, electricity, overhang) exclosures. Experiments to detect *E. rosea* using dogs are ongoing. Ant surveys allowed for the detection and eradication of an incipient population of *Solenopsis geminata*. Black twig borer (*Xylosandrus compactus*) traps are deployed around endangered trees. Rare taxa are responding to these efforts; in 2009, wild seedlings of *C. superba* were documented for the first time in over 30 years.

Island restoration on the Faraday-Ramsay Island group in Gwaii Haanas National Park Reserve and Haida Heritage Site, Haida Gwaii, Canada

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Gwaii Haanas National Park Reserve and Haida Heritage Site is a large protected area jointly managed by the Council of the Haida Nation and Parks Canada Agency. It is located in the southern region of Haida Gwaii, a remote off-shore archipelago of over 150 islands (~1 million hectares) in the Pacific Northwest of Canada. The Gwaii Haanas management plan and State of the Protected Area reports identify introduced species of deer, elk (*Cervus canadensis*), rats (*Rattus* spp.), beavers (*Castor canadensis*), muskrats (*Ondatra zibethicus*), raccoons (*Procyon lotor*), red squirrels (*Tamiasciurus hudsonicus*), house mice (*Mus musculus*), amphibians, birds and many species of invasive plants as the biggest threat to the ecological integrity of Gwaii Haanas. Many introduced species in Gwaii Haanas are widespread throughout the archipelago; however, some island groups have been less impacted because of their relative isolation and limited human use history. Under our mandate to protect and present examples of our natural heritage, the priority to restore these islands is high. In the island group extending from Faraday Island to Ramsay Island, the only species of introduced vertebrates are ship rats (*Rattus rattus*), Norway rats (*R. norvegicus*), and sitka black-tailed deer (*Odocoileus hemionus sitkensis*) in addition to an unknown number of introduced plants occurring at low density along island margins; it is thus an excellent candidate for complete eradication of introduced species. Our Night Birds Returning project endeavours to eradicate introduced rats from seabird nesting islands in this group, while exploring the long-term ecosystem impacts of rat removal, including both direct and indirect impacts to the terrestrial and intertidal areas surrounding these islands. Building on the work of other successful projects, this work is proposed in two stages, starting with a smaller chain of islands (100 ha) to build capacity and community support. Long term plans are under consideration to target deer removal, but logistical difficulties present many challenges. A small scale experimental project to eradicate one invasive plant species is underway, while a larger framework to guide the control and eradication of all introduced plants is being developed.

Population level impacts of localised ferret control: storing up problems for the future?

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Eradication of introduced mammalian predators is not always an immediately feasible option because of logistical, financial and social constraints. Thus, in many cases, lethal control is carried out only around key sites, often with little study of the population level impacts on the controlled species. We studied the behavioural ecology and population dynamics of feral ferrets (*Mustela furo*) on Rathlin Island, UK both pre- and post-control, to examine the effects on the entire island population. Prior to control, over-winter ferret densities were relatively low but animals maintained large home range overlaps and were often found in close association with other individuals. Control was then carried out in limited blocks to mimic protection of important areas for breeding ground-nesting birds. This was highly effective in reducing ferret numbers, with no immigration detected prior to juvenile dispersal. However, the population was found to have substantially increased in the winter following control, remaining high throughout, facilitated by the lack of territoriality. Our study thus suggests limited removal may be counter-productive, and demonstrates how apparently effective control can actually exacerbate the situation in subsequent seasons. This paradox merits further consideration as it may also act for other flexible species, particularly if defining resources such as shelter or food are not limiting.

The Pacific Invasives Initiative Resource Kit for planning rodent and cat eradication projects

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Successful removal of invasive species, such as rodents and cats, from islands requires comprehensive planning. Through its extensive capacity building work with project partners in the Pacific, the Pacific Invasives Initiative (PII) has identified the need for information resources to assist Pacific practitioners in carrying out their invasive species eradication projects. Currently, project managers often do not know where to access relevant information and/or gather information from a variety of sources which can be very time consuming. In response to this, PII has produced a Resource Kit for Planning Rodent and Cat Eradication Projects. The resource kit acts as a "one stop shop" and comprises the PII Development and Implementation Planning Process and all essential supporting tools. The resource kit provides access to a range of information sources including current knowledge and best practice. While the focus of the resource kit is the islands of the Pacific, many of the tools can be readily adapted to other island projects, making it a global capacity building tool. This paper describes the Planning Process and how the resource kit tools will be used to increase the effectiveness of invasive species eradication projects.

The Island Eradication Advisory Group (IEAG) – A model of effective technical support for eradication project planning and management

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The IEAG is a small group of New Zealand Department of Conservation (DOC) staff who represent the best island eradication experience available within DOC. Set up in 1997 to capture existing knowledge and expertise and provide technical advice to up-and-coming DOC projects, the role has diversified into six key areas. These are technical support for eradication projects and island biosecurity; evaluation of best practice for pest eradication; building capability within DOC for pest eradication work; advice on national priorities for island eradication projects; and international networking to maintain DOC's knowledge base by participating in the exploration and resolution of island eradication issues worldwide. Key elements to the success of the group are: a strong customer focus to meet the needs of the project manager; clear separation between advice and decision-making; a team approach to each project; and effective communication. The group meets three times a year and these meetings involve discussion and problem solving with project managers which are then followed up by written advice agreed at the meeting. The IEAG will respond to requests for advice at any time to meet the needs of project managers. Individual members contribute to group discussions via email or conference call to provide a collective view. Many projects have the IEAG undertake pre-operational 'readiness checks' to identify outstanding issues that need to be addressed before implementation. Examples of projects involving IEAG are presented. Key elements in the success of IEAG advice are: robust debate and review involving the IEAG and the project managers; making the most of collective knowledge; challenging assumptions and growing project managers' experience. We think this approach can be adapted to be useful in other parts of the world.

Disperser communities and legacies of goat grazing determine forest succession on the remote Three Kings Islands, New Zealand

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Many remote islands are degraded as a result of deforestation and browsing of vegetation by introduced goats. Goat (*Capra hircus*) eradication is therefore a focus for island restoration but there are few long-term records of changes to islands after eradications. Goats were eradicated from Great Island (Manawa Tawhi), 60 km from the northern tip of New Zealand, in 1946. Three permanent vegetation study plots were established on the island, across a sequence of forest succession, immediately after goat eradication and provide a 57-year record of change. Over the first 17 years, tree diversity in plots increased due to the recruitment of palatable trees. Over the next 40 years, diversity remained similar and forests have been less dynamic. Unpalatable understorey sedges, present when goats were abundant, have persisted and may be impeding tree seedling establishment. Most woody plant species on the island are bird-dispersed. Non-native *Turdus* species are probably important dispersers of many of the small-seeded species. Large-seeded species were unable to germinate away from parents until the native pigeon (*Hemiphaga novaeseelandiae*) were established on the island during the last decade. The slow rate of succession after goat eradication and the current low-diversity forests, compared with the available species pool, reflect legacies of past deforestation, communities induced by goat grazing, and the limited capacity of the resident bird species to disperse many of the potential canopy trees. Our results indicate that restoration of remote islands could require manipulation of goat-induced vegetation or may require sufficient time for favourable habitat for keystone dispersers to develop.

Of rats and birds: creating a seabirds' paradise on Dog Island, Anguilla

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Invasive species are known to cause severe impacts on island ecosystems. One such invasive known to have deleterious effects on islands is the ship rat (*Rattus rattus*). These rats are a potential threat to seabirds. Live traps were utilised to conduct a feasibility study to ascertain the presence of rats on Dog Island, Anguilla, which hosts eight species of seabirds, including one of the Caribbean's largest nesting populations of sooty terns (*Sterna fuscata*) (170,000 pairs). The results indicated that though the ship rat population is relatively high, it should be technically possible to eradicate them from the island using brodifacoum bait and ground-based rat eradication techniques, both of which have been successfully used on other islands. It is anticipated the eradication of ship rats will be achieved within thirteen weeks of the commencement of the programme. It is also expected that the eradication of rats on Dog Island will enhance the island's seabird populations as well as its biodiversity in general.

Developing national eradication capacity for the restoration of globally important seabird islands in the Pacific

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The Pacific island archipelagos of French Polynesia, New Caledonia, Palau, and Fiji support a diverse seabird fauna but, many species and breeding colonies are threatened as a result of the introduction of mammalian predators. Several of these island seabird colonies are globally Important Bird Areas (IBAs) and priorities for conservation. As such, BirdLife International and national non-government conservation organisations in French Polynesia, New Caledonia, Palau, and Fiji implemented a regional island restoration programme between 2007 and 2009 with the aim of eradicating rats from seabird IBAs. How this programme has lead to the development of eradication capacity in four countries, resulting in the completion of rat eradication operations for 16 islands of global importance for seabirds, is discussed, as are the initial results and future restoration priorities and capacity needs.

Toxins, baits and delivery systems for island use

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While there are issues with the repeat use of baits containing brodifacoum in the environment, one-off use for eradication of rodents can result in benefits that significantly outweigh non-target effects. This has been a recommended use pattern for more than 100 islands around the coast of NZ which have been cleared of introduced unwanted rats (*Rattus* spp.) and mice (*Mus musculus*). Nevertheless, difficulties with the existing baits provide a stimulus to search for baits that more effectively target mice as well as rats for island eradication. While alternatives to brodifacoum are seen as more important for enabling effective sustained control, they may, in some situations, still have potential benefits for pest eradication on islands. Current product development is focused on extending the utility of existing "low residue" toxins such as zinc phosphide, cholecalciferol and a combination of coumatetralyl and cholecalciferol in baits that are particularly palatable to rats and mice. We are also pursuing the registration of products containing substances such as sodium nitrite and para-aminopropiophenone (PAPP) and are working on baits and delivery systems to improve target specificity. Our work with PAPP for stoat (*Mustela erminea*) and cat (*Felis catus*) control in NZ provides a platform to search for a novel class of rodenticides but this will take a few years to complete. In the short term diphacinone, cholecalciferol and low dose cholecalciferol in combination with coumatetralyl represent low risk acute toxins for control of rats and mice without secondary poisoning. Research focusing on the registration of new solid multispecies baits should yield registered alternative rodenticide baits suitable for aerial application.

Estimating spatio-temporal change in population size of an invasive species from capture records: application for the mongoose eradication project on Amami Island, Japan

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Estimation of the effect of the control and the spatio-temporal change in the population size of an invasive alien species helps to evaluate and improve the strategy for the eradication. It is necessary to establish models to estimate the population dynamics of an invasive alien species from the information obtained in the eradication process. On Amami Island, Japan, small Asian mongoose (*Herpestes javanicus*) was introduced as a biological control agent for the native poisonous snake, habu (*Protobothrops flavoviridis*), in 1979. The predation of the non-target endemic animals by the mongoose has been a great threat of the biodiversity conservation. In 2000, the Ministry of Environment began an eradication project against mongoose. The removal of the mongoose has been done using traps, and the location and capture history of almost all the traps have been recorded. In this study, we established a hierarchical model to estimate the efficiency of capture and the spatio-temporal change in the population size from the capture history. Our model consists of the population dynamics and the relationship of the population size and the trapping effort to the number of capture. Our model allows the spatio-temporal heterogeneity in the population growth rate. Using Markov Chain Monte Carlo (MCMC) method, the population size and its growth rate in each time and place and the capture probability of the trap were estimated from the data of the number of captured mongoose and the trapping effort. We also suggested the index of the optimal spatial arrangement of traps from the estimated values. The data used in this study was obtained by Amami Mongoose Eradication Project by Naha Nature Conservation Office, Ministry of the Environment, Japan. Trapping and the recording of captures have been done by Amami Mongoose Busters.

The origin of amphibian chytridiomycosis: did it come from Japan?

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A serious disease of amphibians caused by the chytrid fungus *Batrachochytrium dendrobatidis* was first discovered in Japan in December 2006 in imported pet frogs. This was the first report of chytridiomycosis in Asia. To inspect the origin and expansion process of the chytrid fungus in Japan, we surveyed the distribution and genetic variation of the fungus among captive and wild frog populations. We established a nested PCR assay that uses two pairs of PCR primers to amplify the internal transcribed spacer (ITS) region of a ribosomal RNA cassette to detect mild fungal infections from as little as 0.001 pg (1 fg) of *B. dendrobatidis* DNA. We collected swab samples from 559 captive amphibians, and 5565 wild amphibians collected at field sites from northern to southwestern Japan. We detected infections in native and exotic species, both in captivity and in the field. Sequencing of PCR products revealed 50 haplotypes of the *B. dendrobatidis* ITS region. Phylogenetic analysis for the haplotypes combined with haplotype sequences already detected in other countries showed that genetic diversity of *Bd* in Japan was higher than that in other countries. Furthermore, it was suggested that three of the haplotypes detected in Japan were specific to the Japanese giant salamander (*Andrias japonicus*) and appeared to have established a commensal relationship with this native amphibian. The highest genetic diversity of *B. dendrobatidis* was found in the sword-tail newt (*Cynops ensicauda*), endemic to Okinawa Islands and the next highest in the alien American bullfrog (*Rana catesbeiana*). From these results, combined with no evidence of chytridiomycosis occurrence in the Japanese native species, we came to a new hypothesis for the source of the fungus: “Asia or Japan origin hypothesis”. To improve chytridiomycosis risk management in the world, we must restrict the amphibian trade, especially from Japan.

Establishing the raccoon control system and its issues in Hokkaido, Japan

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Pet raccoons (*Procyon lotor*) that have been abandoned or escaped have become established in Japan and are extending their range, damaging agriculture nationwide and substantially impacting on the native ecosystem. In Hokkaido, scientists and governments have been addressing this issue together. Initially, raccoons were captured as part of harmful wildlife control; however, this approach lacked evaluation of the captures. Consequently, to contribute to consensus building for the control system, target capture numbers were determined by predicting population dynamics scientifically with reproduction data analysis of captured individuals. We set model areas and verified the efficacy of the capture by continuing the same capture approach. As a result, it was shown that population density can be kept at, or below, two animals/km^2 solely by placing cage traps every 500 m in the area and conducting three continuous weeks of capture once a year. Also, as there was a correlation between the population density and the capture per unit effort (CPUE), CPUE was introduced as a relative index of population density. At present, local governments aim to reduce CPUE to 1 animal/100 trap nights, corresponding approximately to a population density of 1 animal/km^2. However, the current capturing method using cage traps is not cost effective in low population density areas. Thus, development of effective capturing approaches in such areas, including training of raccoon detective dogs, is a challenge. Furthermore, although Japan is deeply concerned about the impact of alien species on the population, it remains relatively unaware of their impact on the native ecosystem. Japanese people have a strong reluctance to kill animals and, therefore, public awareness-raising is also required, as well as the reinforcement of social education regarding invasive alien species.

The invasion of the Argentine ant across continents, and their eradication

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The Argentine ant (*Linepithema humile*) has successfully spread from its native range in South America across much of the globe. This species is highly polygynous and possesses a social structure, called ‘supercolony’ whereby individuals mix freely among separated nests. The introduced populations of the Argentine ant are characterised by the formation of very large supercolonies across thousands of kilometres, whereas colony size is generally smaller in the native ranges. Gene flow among supercolonies has been considered to be very limited or even absent. The Argentine ant, first noted in 1993, is now found in several regions of Japan. Early detection, as well as rapid control, is required to prevent further expansion of the species. A vital component of this prevention is the identification of pathways of introduction into new locations. First, we attempted to demonstrate the genetic structure of the Argentine ant to understand its dispersal history. Sequencing of the mitochondrial DNA from the Japanese and overseas populations showed that one haplotype is shared among different populations distributed in USA, Europe, Australia, and Japan. Three haplotypes were shared among four supercolonies with high levels of aggression in Japan. These results indicate that one massive supercolony is distributed across the continent and that replicated introductions may occur in Japan. Secondly, for understanding the mechanism of formation of the massive supercolony we examined whether gene flow can occur among supercolonies. As a result of investigations of reproductive schedules and aggression of workers toward males, gene flow may be limited between adjacent supercolonies. Finally, we introduce the eradication trials of the Argentine ant in Japan.

Mongoose, rat and acorn - forest dynamics and ecosystem management on Amami Island, Japan

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The small Indian mongoose (*Hepestes javanicus*) has been spreading on Amami Island for 30 years. The island is at the north-eastern most corner of the Oriental region, and is rich in endemic species of subtropical forest. In the last decade, there has been intensive control of mongoose by the Ministry of the Environment. Mongoose are at low density and distribution covers up to 300 square kilometres over a complex forest ecosystem with complicated terrain. The island hosts another invasive alien mammal, ship rat (*Rattus rattus*), which greatly increases in abundance in forest after the rich acorn crops of the ever-green oak (*Castanopsis sieboldii*), an extremely widespread tree. Ship rats and a wintering thrush (*Turdus pallidus*) are two important winter foods for mongoose. Reproduction, dispersal, and also trapping performance of mongoose should depend on the abundance of rats, the thrush and other animals, which also fluctuate with acorn production. Understanding the patterns and process of the food web through acorns, rats, other native animals, and mongoose helps with developing optimal control strategies (lowest cost and highest benefit) and to investigate the possibility of mongoose eradication. Ecosystem management thinking is thus indispensable for invasive species control on Amami Island.

Eradication of exotic rodents off six high conservation value Western Australian islands

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Introduced rodents are a major threat to the biodiversity of islands around the world, including Australia. In 2009, a Threat Abatement Plan to reduce the impacts of exotic rodents on biodiversity on Australian offshore islands of less than 100,000 hectares was approved by the Commonwealth Government. Introduced rodents are known from at least 69 islands off the Western Australia coast and since the 1980s successful eradication programmes have been implemented on half of these. This project will eradicate introduced house mice (*Mus musculus*) and rats (*Rattus* spp) from another six high conservation value islands over a four year period. House mice will be eradicated from Three Bays and Faure islands in Shark Bay; ship rats (*R. rattus*) from Sunday and Long in the Kimberley and Direction Island in the Cocos-Keeling group; and Pacific rats (*R. exulans*) from Adele Island also off the Kimberley coast. In addition, a survey of Dirk Hartog Island in the Shark Bay World Heritage Area will be undertaken to confirm or otherwise, the presence of ship rats on this 68,000 ha island. Where bait spreading by helicopter is not practical and where non-target issues are present, appropriate bait stations will be developed and deployed. Where bait stations can not be developed to prevent access to baits by non-target species, some may be removed from the island, eradication undertaken and the non-targets returned once eradication has been confirmed. Eradication will most likely be by baiting with the anticoagulant poison, brodifacoum; however, there have been recent developments with other baiting formulations and these will be utilised if appropriate. The eradication programmes will be supported by short and longer term monitoring programmes, an education programme and quarantine protocols will be developed to ensure islands remain free of introduced rodents. Indigenous communities will be engaged to assist with eradication and monitoring activities.

Effectiveness of bait tubes for brown treesnake control on Guam

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A bait tube is a device with which a toxicant inserted in a dead mouse (*Mus musculus*) can be delivered to invasive brown treesnakes (*Boiga irregularis*) with low risk of non-target bait take. We tested two bait tube designs in a 5ha snake enclosure where the identity of virtually every snake is known. Instead of using toxicants, we implanted radio transmitters in small (6.6±1.4 g) and large (21.8±2.9 g) bait mice. Knowing all snakes present in the population allowed us to characterize not only covariates of snakes taking bait, but also those of snakes evading our mock control effort, and if snake covariates interacted with any design variable in determining targeting rate. Tube design had no effect on take rate. Snake snout-vent length was a strong predictor of success: none of the 29 snakes smaller than 843 mm took any bait, while the 126 snakes ≥843 mm were responsible for a total of 164 bait takes. The smallest of these snakes were able to ingest small and large mice, but tended to consume small bait at a higher rate than large bait. The main reason for our failure to target smallest snakes appears not to be gape limitation, but rather that small snakes prefer other prey (lizards). The time it takes a snake to grow from the size threshold observed to the size of maturation has implications for the interval between discrete efforts using toxic bait. Targeting all snakes before reproduction can occur is highly desirable; otherwise, a new cohort of refractory snakes may enter the population.

Economics of biocontrol for management of *Miconia calvescens*

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Ecological devastation in Tahiti and the threat to biodiversity and watersheds in Hawaii has deemed *Miconia calvescens* a priority invasive plant. Since the early 1990s, millions of dollars have been spent on advanced technologies and best management practices to reduce the prevalence and spread of the tree. On the islands of Hawaii and Maui, aerial reconnaissance and GIS are used to monitor and map populations; manual removal and herbicide treatments are used to destroy the plants. Long term suppression of *Miconia* remains at bay, and years of effort is being continually threatened by rising costs and uncertain budgets. To this end, scientists in Hawaii have been collecting and testing biological control agents for their effectiveness and host specificity. Using information from Hawaii tests of a stem weevil *Cryptorhynchus melastomae* and a nematode *Ditylenchus gallaeformis*, we simulated release scenarios; estimated the total cost of research, development, release, and monitoring; and compared biocontrol costs to projected expenditures under current best management practices. We estimated that net benefits from biocontrol agent release on a *single* Hawaiian island could reach US$10 million in 50 years. These results strongly indicate that continued research for a safe and effective biocontrol agent in Hawaii is economically warranted.

Eradicating foxes from Phillip Island, Victoria: techniques used and ecological implications

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The red fox (*Vulpes vulpes*) is considered to be the greatest land-based threat to little penguins (*Eudyptula minor*) on Phillip Island, in Victoria, Australia. Phillip Island Nature Parks has commissioned a fox eradication strategy to manage the threat and is committed to eradicating foxes from Phillip Island within five years. Island-wide 1080 baiting on private and public land has been employed as the most efficient method for broad-scale control and is supplemented by other methods, such as trapping, spotlighting and den fumigation. The use of scent dogs to detect fox scats is seen to be instrumental in locating and destroying the last few individual foxes on Phillip Island. As foxes are a cryptic species, monitoring fox abundance is difficult. Deriving relative abundance indices from a number of different parameters influenced by fox presence is considered the best way to assess success of the eradication programme. The number of penguins killed by foxes has fallen to extremely low levels (two penguins in 2008/2009 from over 300 penguins in previous years) and other key indicator species such as Cape Barren geese (*Cereopsis novaehollandiae*) and masked lapwings (*Vanellus miles*) are showing signs of population increases. Comparing bait take, spotlight transects and efficiency or catch per unit effort (CPUE) of each method over time is another method to gauge the success of the programme. Another result of the eradication programme has been an increase in mesopredators such as feral cats due to reduced competition and direct predation from foxes. Nature Park staff destroyed over 130 feral cats from farmland and reserves on Phillip Island last year and are now undertaking a public education campaign to educate the community on responsible cat ownership and the threat cats pose to native wildlife.

Goat eradication on Kangaroo Island, South Australia

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The high conservation value of Kangaroo Island has prompted the KI Natural Resources Management Board, in association with the Invasive Animals Cooperative Research Centre and South Australian Government, to implement a feral animal control programme targeting a number of species. Eradication of goats (*Capra hircus*) is one of the most successful components of the programme. Goats arrived with the first settlers to Kangaroo Island nearly 200 years ago and over the years the western and northern coastal environments have become population strongholds. Goats were controlled by opportunistic ground shooting until a coordinated strategic approach was set in place in 2006. Public meetings and discussions with the community helped identify the area of the island populated by feral goats. That area was divided into seven management units (MUs) using natural barriers as boundaries to help systematic eradication, one MU at a time, and limit re-infestations. Sterilised goats fitted with radio-telemetry collars (Judas goats) were first released into the first three MUs to join feral populations and determine effectiveness in this environment. Over the past three years, the 27 Judas goats released have provided information on movements, including the location of watering points and shelter locations, group size and behaviour in specific areas. Because of the Judas goats, 997 feral goats have been easily found and destroyed with little extra effort required for the last few. Four management units are now in a monitoring stage with no feral goats spotted for over a year. The remaining three management units are currently being targeted and eradication should be complete by 2012. The success of the programme is attributed to the well-planned approach, effective destruction techniques implemented by skilled staff, and the support and participation of all stakeholders.

Eradication of non-native tilapia from a natural crater lake in the Galapagos Archipelago, Ecuador

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In 2006, Nile tilapia (*Oreochromis niloticus*) were discovered in Laguna El Junco, a natural crater lake on Isla San Cristobal, Galapagos Archipelago, Ecuador. The largest body of freshwater in the Galapagos, El Junco was naturally devoid of fishes. Galapagos National Park, in conjunction with the Charles Darwin Foundation, drafted a plan in 2006 proposing application of rotenone, a commonly used fish poison, to eradicate the tilapia. In August 2007, we visited the lake and surveyed surrounding areas. We verified the identity of the fish, confirmed that the lake population was reproducing, and concluded the tilapia were likely restricted to the lake. Eradication was justified because predation by tilapia was changing the composition and abundance of the lake's native invertebrate community, negatively affecting some species considered endemic to the Galapagos. Moreover, the longer the tilapia persisted, the greater the likelihood of dispersal into other aquatic habitats. We conducted a series of toxicological tests on tilapia and invertebrates from El Junco to determine the optimal concentrations of rotenone to apply. We also sampled aquatic invertebrates from the lake, reserving some in refuge tanks for later restocking. Following months of planning, on 25 January 2008 liquid rotenone was applied and over the next few days approximately 40,000 dead and dying tilapia were removed from the lake. After tilapia removal, and once all residual rotenone in the lake had degraded sufficiently, captive invertebrates were released back into El Junco to speed recovery of invertebrate communities. No live tilapia have been collected or observed since 31 January 2008.

A newly recorded alien population of a lizard *Plestiodon japonicus* in Hachijojima Island, central Japan

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The scincid lizard *Plestiodon japonicus* is naturally distributed in part of the Japanese main islands and the coastal region of eastern Russia. Since the spring of 2004, an alien population of this species has been recorded in Hachijojima Island, central Japan, where a congeneric population of *P. latiscutatus* is naturally distributed. As native lizard populations are already facing extinction from an alien predator (the Japanese weasel (*Mustela itatsi*)), the alien lizard population will elevate the extinction risk of the native species through competition and introgressive hybridisation. Our preliminary study in 2007 and 2008 suggested the following: the alien species has already established a breeding population; the alien population was localised in a small part of the island and did not occur alongside the native congener; the alien population had slight genetic variation and therefore seemed to originate from a single source; little or no hybridisation with the native congener occurred. The invasion of the alien population may be at an early stage and therefore prompt eradication will suppress the impacts of the alien lizard.

Context matters: assessing the biodiversity benefits of pest eradication

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The biodiversity benefit obtained from the eradication of a particular pest on a particular island depends on both the biodiversity context around the island, as well as the pest context (e.g., suite of pests) on the island. The biodiversity benefits of pest eradication on an island depend on the archipelagic biodiversity context, including the rarity of the native species on the island, whether they are present on other islands, and whether they are being managed on other islands. Well-known concepts from conservation planning, such as complementarity and irreplaceability, can be used to illustrate the importance of archipelagic biodiversity context in choosing what pests to eradicate on which islands. Pest context matters also; the marginal benefits of removing a particular pest depends not only on the effects of that pest on the native species, but also on what other pests are on the island, and their effects on native species. I illustrate both of these contextual effects, using the Vital Sites model, which contains spatially explicit information on the New Zealand distributions of native species and pests, and simple models of the impacts of pests on native species. The removal of a particular pest provides more biodiversity benefit if it is the last pest removed from the island, than if it is the first pest removed from the island. This result is independent from, and exacerbated by, increases in the density of remaining pests due to reduced competition or predation from the removed pest. Furthermore, the marginal operational costs of controlling a particular pest are likely to decrease, as more pests are controlled. Both of these effects argue for multiple (rather than single) pest eradications. These results have important consequences for deciding what pests to eradicate or control on which islands, and whether to do single or multiple eradications.

Control of the invasive ship rat (*Rattus rattus*) and Pacific rat (*Rattus exulans*) using a large scale trapping grid for endangered tree snail and plant conservation in Hawaii

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Introduced rats (*Rattus* sp.) in Hawaii are known predators of birds, tree snails, and plants. Since 1997, the Oahu Army Natural Resources Program has been controlling rats through the use of diphacinone rodenticide in bait stations and snap traps on a relatively small scale at multiple sites for the protection of the endangered Oahu elepaio (*Chasempis sandwichensis ibidis*), five endangered Oahu tree snail species (*Achatinella* sp.), and seven endangered plants species. In May 2009, rat control was initiated over a 26 ha forested management unit with 400 snap traps on the island of Oahu. The New Zealand Department of Conservation current best practice rat trap technology is being utilised for the first time in Hawaii with this trapping effort. Rat activity within the management unit will be monitored through the use of tracking tunnels. Forest health, the endangered plant *Cyanea superba* subsp. *superba*, the Oahu tree snail *Achatinella mustelina*, and native invertebrates will be monitored closely to determine the effectiveness of the methodology. Introduced slugs and the predator snail *Euglandia rosea* will also be monitored to determine whether rats are suppressing these two highly invasive species.

Aerial baiting for rodent eradication programmes

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Aerial spread of cereal baits containing brodifacoum is the primary technique employed for rodent eradication programmes on islands. More recently, with the approval of a Code of Practice, this technique has been expanded to mainland sites surrounded by pest proof fences. Skywork Helicopters has put a considerable investment into the development and refinement of gear and equipment for aerial baiting. This is based on a system of continuous improvement and experience working on mainland sites such as Tawharanui Regional Park (Northland), Rotokare Scenic Reserve (Taranaki) and offshore islands including Little Barrier, Macauley Island, Rangitoto/Motutapu, Great Barrier and the Kaikoura Island chain. Aerial baiting operations for eradication programmes require exacting standards and the use of experienced pilots and ground crew. These operations are often conducted in remote environments and require effective logistical support and good problem solving skills. Planning and operational management of these operations requires good knowledge of the pest species present, the land area the operation is to be undertaken in, as well as factors that may influence a successful baiting operation, such as weather, steep cliffs, accuracy of helicopter buckets, and the use of DGPS navigational systems.

An attempt at a surveillance sensitivity comparison in Amami-ohshima Island, Japan

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Many endemic species in Japan, especially on small islands, are now threatened by invasive alien species. In 1979, the small Asian mongoose (*Herpestes javanicus*) was introduced to Amami-ohshima Island to control native poisonous habu snakes (*Protobothrops flavoviridis*). However, the mongoose has had a predatory impact on endemic animals. From 2005, the Ministry of the Environment began a 10 year project to eradicate the mongoose from the island. This has successfully decreased population density of the mongoose. Some scientists (e.g., John Parkes and Alan Saunders) gave advice about this project at the Symposium of Control Strategy of Invasive Alien Mammals 2008, held in Okinawa Japan. They advised that at the next stage, we should use the capture technique in the low density area, as well as a method to investigate the presence or absence of the mongoose. Responding to their advice, we plan to develop some methods to investigate the presence or absence of the mongoose. To do this, we need to know the relationship between known frequency and population density. At first, we will research to find a relationship between frequency, using a sensor camera and population density as it is thought that the photographed frequency is proportional to population density. We report on the design and the progress of our research.

Trap allocation strategy for the mongoose eradication project on Amami-Ohshima Island, Japan

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When the establishment of an invasive alien species has once been detected, we should take appropriate steps such as eradication, containment, or control. If eradication is not feasible, the goal of control is to maintain reduced population sizes or to prevent expansion of the distribution of the invasive species. There is a trade-off between high and low population density areas in population control. When the project manager allocates many traps in a center of distribution, the population may continue to expand from the margins. Eradication is only possible if spatial trap allocation is appropriate. In many cases, the project manager does not have sufficient information about the distribution of the target species. Therefore, trap allocation based on the capture results from the previous year is probably useful to control the target species. We examined effective trap allocation by using a lattice model in both cases whether eradication is possible or impossible. We suggest an effective trap allocation strategy using parameter values of small Indian mongoose (*Herpestes javanicus*) on Amami-Ohshima Island where a mongoose eradication project has been carried out by the Ministry of the Environment.

Canine detection of free-ranging brown treesnakes on Guam

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We investigated canine teams (dogs (*Canis familiaris*) and their handlers) on Guam as a potential tool for finding exotic brown treesnakes (*Boiga irregularis*) in the wild. Canine teams searched a defined 40m × 40m forested area with a snake that had consumed a dead mouse (*Mus musculus*) containing a radio-transmitter. To avoid tainting the target with human scent, no snake was handled prior to searches. Trials were conducted during the morning, when snakes were usually hidden in refugia. A tracker knew the snake's location, but dog handlers and data recorders did not. Of 85 trials conducted over 4 months, the two canine teams had an average success rate of 35% of correctly defining a 5m square area that contained the transmittered snake; the team with the most training had a success rate of 44% compared with 26% for the newer team. Eleven sheds from wild snakes were found and, although dogs alerted outside the location of transmittered snakes, only one wild, non-transmittered snake was found during the trials, possibly reflecting the difficulty humans have in locating snakes in refugia. We evaluated success at finding snakes as a function of canine team, time, canine success at the previous trial (we predicted that dogs that had been recently rewarded might be more successful), environmental conditions, cloud coverage, average humidity, average temperature, average wind speed, rain during trial, and rain in previous 6 hours), snake perch height, and snake characteristics (snout-vent length and sex). Success rate increased over the course of the trials, perhaps due to increased searching experience. Canine team success also increased with increasing average humidity and decreased with increasing average wind speed. Our results suggest that dogs could be useful at detecting snakes in refugia, but techniques are needed to help humans pinpoint a snake's location once a dog has alerted.

Restoration of globally important seabird islands in Fiji by the removal of rats

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Seabirds are becoming increasingly scarce among the more than 300 islands of the Fijian archipelago. Several reasons have been attributed to this. Key amongst these are the introduction of alien mammals to breeding islands, particularly rats (*Rattus* spp), feral cats (*Felis catus*), pigs (*Sus scrofa*) and dogs (*Canis familiaris*) and other anthropogenic influences such as fire and harvesting. In an effort to protect breeding seabird colonies in Fiji, BirdLife International Fiji Programme undertook an assessment of seabird islands identifying sites of national and global importance. Threat assessments confirmed the presence of at least one species of introduced mammalian predator on all islands. In 2006, following its identification as an Important Bird Area, Vatu-i-ra Island was subject to a Pacific rat (*Rattus exulans*) eradication operation, to remove the only invasive predator of seabirds from the island. This operation was a success and in 2008 was followed by rodent eradications from seven of the Ringgold Islands and Mabualau. Community consultation is a vital component to invasive species management in Fiji, as 75% of the land tenure is native owned. The development and implementation of these projects has been conducted using a participatory process where capacity development has been extended to landowning communities. Despite the achievements and local support for the restoration of seabird islands, the biggest challenge remains with the long term management and maintenance of pest free islands. The current approach to this is presented.

Management of the red crab (*Gecarcoidea natalis*) on Christmas Island, Indian Ocean: the efficacy of a yellow crazy ant (*Anoplolepis gracilipes*) baiting programme

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Christmas Island is located approximately 360km south of the western head of Java, Indonesia. One major biological feature of the island is the unusually high density of red crabs (*Gecarcoidea natalis*), which are considered a 'keystone species'. *Gecarcoidea natalis* can determine vegetation communities through their herbivory and limit the potential for colonisation by some introduced species. In the late 1990s, *G. natalis* was extirpated from large areas of Christmas Island after the formation of supercolonies by the introduced yellow crazy ant (*Anoplolepis gracilipes*). In response, Christmas Island National Parks embarked on a YCA supercolony baiting programme that has been running continually since 2001. Here we report on the outcomes of a biannual island-wide survey that has now been conducted five times to monitor changes in crab burrow densities relative to ant baiting. On each survey, occupied *G. natalis* burrows are counted along a 50 metre transect at 877 survey points across the island. We used a Bayesian hierarchical spatial model to show that despite the death of up to 33% of *G. natalis* in the early phase of ant supercolony formation, densities of crab burrows have remained stable since 2001. However, significant, but more localised changes in burrow densities occur on a regular basis, suggesting a dynamic system.

Improving "internal" biosecurity in the Falkland Islands: a pragmatic approach

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The Falkland Islands are an archipelago of more than 700 islands, with a wide range of sizes, topography, and ownership arrangements. Many islands are privately owned: some of these are farmed, and some have residences that are occupied for all or part of the year; other islands are uninhabited and treated as reserves or used for grazing livestock. The main transport methods between islands are private boats, a ferry, the Falkland Islands Government Air Service (light aircraft) and helicopters. There are three species of rodents present on the Falkland Islands: ship rats (*Rattus rattus*), Norwegian rats (*R. norvegicus*) and house mice (*Mus musculus*). Some islands have remained rodent-free, but many have one or two of these species present. Since 2001, a successful programme of rodent eradications has been undertaken on the Islands, with more than 20 islands cleared. With increasing numbers of rodent-free islands, reducing the risk of reinvasion (or new invasion) has become a growing priority. The recent emergence of new pest species has also raised the profile of biosecurity issues amongst landowners, the general public and the Falkland Islands Government. A pragmatic, non-regulatory approach has been taken to improve "internal" or inter-island biosecurity on the Falkland Islands in the last three years. Current worldwide best practices were investigated and elements from different programmes were selected to create a system of island biosecurity that would be manageable, cost-effective, and achievable for different landowners and users. This approach has involved improving public awareness and education rather than introducing legal regulations. This approach could be applied to other island groups where reducing the risk of pest invasion is important but legal regulations are lacking. This work was funded by the European Union's EDF-9 fund and the Falkland Islands Government, with support from island owners and Falklands Conservation.

When failure is not an option: applying new tools to rodent eradication planning

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Rodent eradication is often successfully used to protect native island biota from the negative impact of introduced rodents. However, as this tool is increasingly applied worldwide, standard eradication methodologies are being challenged by increasingly complex systems, e.g., commensal rodents, multi-island atolls and tropical ecosystems. To address these issues, Island Conservation applied three new tools to refine rodent eradication planning: a biomarker bait; hand-broadcast, using GIS; and genetic sampling protocols. On Palmyra Atoll and Wake Island (tropical Pacific) and Desecheo Island (Caribbean), a placebo bait, using the biomarker pyranine (a fluorescent dye), was used to determine bait application rates for high density *Rattus* sp. and commensal rodent populations, and to track bait consumption by land crabs and other invertebrate consumers, which are potential secondary sources of rodenticide for non-target predators. On Wake Island, placebo bait was hand-broadcast across 10 ha study plots using hand-held GPS units uploaded with a GIS layer of predefined points at which bait was broadcast. In the event of a rodent eradication failure, Island Conservation has also developed protocols for genetic sampling of rodent populations to determine if failure was due to re-emergence of a residual population or re-invasion from an outside source. Together, these tools have improved our efficiency of ground-based bait application, enabled a better understanding of non-target bait consumption, and overall have improved our rodent eradication planning, including learning from potential failures.

Community-based nutria control by traditional irrigation systems

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Nutria (*Myocastor coypus*) was originally farmed for its fur, but, since being abandoned, has naturalised in Honshu island, Japan. Especially in and around the agricultural regions of Hyogo Prefecture, this semi-aquatic mammal has rapidly expanded its distribution range in the last decade. From observations in Kasai City it was discovered that the nutria exhausted some aquatic plants and were threatening some invertebrate species, including an endangered Japanese dragonfly, *Libellula angelina*. Although nutria has been recognised as a serious invasive species in wetland ecosystems, it is difficult to eliminate them by hunting or trapping because their home ranges are within villages. To develop an ecological method to control the nutria population, we investigated their dispersal pattern and attempted to exclude them from some drainage systems using an irrigation technique in Kasai City. At first, from the analysis of records, we clarified that the nutria dispersed through non-manipulated irrigation canal systems and bred in the banks of ponds. We then contrasted their utilisation of canals and banks between manipulated and non-manipulated systems, and it was confirmed that nutria avoided fast currents as well as large fluctuations in water levels, probably because of their difficulties in moving and nesting. We tried to alter the water level and volumes of un-manipulated irrigation systems, mainly in winter season, and observed the movements of nutria. As a result they rarely moved from the lower to upper reaches. Nesting female groups abandoned their upriver nests and vegetation started to recover in the following year. In conclusion, reactivation of this old-style indigenous irrigation system is an effective and receptive (a community-based) method to control nutria and to restore the specific wetland ecosystem.

Accomplishments and impact of the NGO, Island Conservation, over 15 years (1994 – 2009)

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Since its inception in 1994, the NGO, Island Conservation, has removed 54 populations of 10 invasive vertebrates from 35 islands totalling >52,000ha. These actions have helped protect 233 populations of 181 insular endemic species and subspecies of plants and vertebrates and 288 populations of 54 species and subspecies of seabirds from the threat of local and global extinction. There were no reinvasions. One eradication attempt failed. These conservation actions and their apparent biodiversity impact demonstrate the potential of private organisations to protect biodiversity by eradicating invasive species from islands.

Snap-trapping, a viable alternative to ground-based poison operations for eradication and/or control of rats in island and mainland situations

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During development of the novel Ka Mate reverse-bait snap-trap, in 2009 450 of the traps were deployed over 75 ha of mature broadleaf/podocarp/kauri forest in Waiaro Sanctuary (Coromandel, New Zealand). The trials were designed to replicate with traps ground-based poison campaigns (e.g., the landmark 1988 eradication of rats from Breaksea Island), and test whether it was possible to achieve similar outcomes without the use of toxins. In Waiaro, rat-catch reduced significantly from 117 *Rattus rattus* killed on night one to less than fifty per check a week later. At six months, catches of 2-10 rats per check were only on the peripheral trap-lines, with no rat incursion or rat sign found within the core of the trapped area for more than three months. More than 800 rats were removed from the Sanctuary, all of which were clean-kill head-strikes. Despite the traps being set in open situations without protective stations, a few mice were the only non-target by-catch. The deployment, effectiveness and problems encountered with various trapping regimes, using a mix of trap types in programmes from wide-ranging localities and habitats worldwide (Seychelles Islands, New Caledonia, Wake Atoll, Hawaii, and several New Zealand sites) are also discussed.

The infection risk and pathogenicity of chytrid fungus *Batrachochytrium dendrobatidis* carried by the Japanese sword tailed newt

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Amphibian chytridiomycosis, caused by the chytrid fungus *Batrachochytrium dendrobatidis*, is a highly virulent disease of amphibians and is known to be a major driver of amphibian declines observed all over the world. In Japan, this fungus was first found in December 2006 from imported pet frogs. The nationwide investigation to assess the risk of pandemic chytridiomycosis to Japanese frogs elucidated that this fungus is distributed all over the Japanese main islands and that the genetic diversity of Japanese chytrid fungus, including more than 30 haplotypes, is much higher than those of fungus in other countries. Thus, several researchers currently consider that Japanese islands are one of the native localities of this fungus and that amphibian chytridiomycosis observed elsewhere in the world might be caused by the fungus derived from Japan. To verify this "Chytridiomycosis out of Asia hypothesis", we surveyed the infection risk and pathogenicity of the chytrid fungus carried by Japanese amphibians. In experimental infection, the chytrid fungus carried by Japanese sword tailed newt (*Cynops ensicauda*) infected South American horned frog (*Ceratophrys ornata*). All frogs infected by Japanese chytrid fungus showed an onset of amphibian chytridiomycosis. Given that Japanese amphibians, including sword tailed newt, frequently have been exported to foreign countries as pets, we must consider that the chytrid fungus carried by Japanese amphibians would also be introduced to foreign countries leading to amphibian chytridiomycosis of native species in host areas.

Coordination mechanisms for invasive species action in the Pacific

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Islands are exceptionally vulnerable to invasive species impacts, but small island nations often do not have the human or financial resources to tackle these threats adequately by themselves, especially projects with heavy one-off costs such as eradications. Pacific nations and territories have a long history of cooperation to enable them to overcome such limitations. Mechanisms and tools have been established to promote collaboration and effective action against invasives in the Pacific, which can serve as models for elsewhere, particularly other oceanic regions. The Pacific Invasives Partnership promotes coordinated prioritisation and assistance from regional and international agencies to countries and territories of the region. Its members include regional intergovernmental agencies, NGOs and other organisations working on invasives issues in more than one Pacific country or territory. The partnership is supported by two regional initiatives: the Pacific Invasives Learning Network, which is a professional aid network for invasive species workers in Pacific countries and territories to facilitate collaboration and exchange of information and skills; and the Pacific Invasives Initiative, which provides assistance with project development, training and links to expertise. These programmes help build local capacity in different ways. A guiding strategy, the Guidelines for Invasive Species Management in the Pacific forms a framework for action by all of them, in which eradication is emphasised as the preferred management objective for established invasives when feasible. The overall goal of these regional initiatives is to assist Pacific island countries and territories in planning and achieving more effective invasive species management.

Dogs working for conservation

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Dogs (*Canis familiaris*) have assisted with mammal eradications in New Zealand for the last 30 years. Since 2002, the Department of Conservation has run a dedicated predator detection dog programme providing dog and handler training and certification, systems development and improvement, a breeding programme and operational support. The dogs are trained to detect the presence of mammalian predators and browsers, including rodents, mustelids, cats (*Felis catus)* and rabbits (*Oryctolagus cuniculus)* for the purposes of audit, incursion contingency response, surveillance, biosecurity quarantine and optimising trap placement. Dogs have proved to be an extremely effective tool for confirming presence when predator numbers are low and other predator detection methods (tracking tunnels, traps, gnaw sticks) are less efficient. Once detected by dogs, the predators are killed using pesticides, traps, or shooting. Since the programme started, these dogs have been involved in many successful pest eradication programmes on islands. The NZ dog programme has also provided international advice, training, and dogs (practical support) for eradication programmes e.g., Macquarie Island, Australia (rabbits) and Amami Island, Japan (mongoose, *Herpestes javanicus*). This paper presents the dog programme and illustrates case studies where use of the dog programme has assisted eradications, including: Raoul Island (cats), Campbell Island (cats and Norway rats, *Rattus norvegicus*), Secretary Island (stoats, *Mustela erminea*), Te Kakahu/ Chalky Island (stoats), Tuhua/Mayor Island (cats), and many contingencies including Motuihe Island where the rodent dog detected the rat within 48 hours of tracks being discovered on tracking cards.

Risk analysis of potential freshwater nuisance fish and other species associated with increased U.S. military presence in Guam and circum-Pacific islands

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The islands of Micronesia have low taxonomic richness of strictly freshwater aquatic species, yet endemism on single islands or island groups is often high. In contrast, non-native aquatic organisms have become increasingly common. Although published reports differ in total numbers, approximately 70-90 species of fish have been introduced into fresh (and some brackish) waters of the western Pacific and Hawaiian islands. In addition to fish, non-indigenous freshwater molluscs and crustaceans have also been introduced. Sources of introductions vary from some that were intentional (e.g., for aquaculture, ornamental trade, mosquito control), to those of accidental or unknown pathways. The ecological and economic effects of these introductions are poorly understood and generally have not been quantified. The U.S. Department of Defense (DoD) projects manifold military operations in the western Pacific, centred in Guam and the Commonwealth of Northern Mariana Islands. Increased traffic of cargo and personnel associated with the expansion of military operations poses elevated risk of the transport of invasive species throughout the region. Consequently, freshwater systems of Micronesian islands and their vertebrate and invertebrate faunas are in need of greater study to determine the extent of threats to the native biota. This project provides a freshwater component to a multi-agency and multi-disciplinary endeavour to evaluate control and management protocols for existing and potential invasive species, as part of a collaborative process to prepare a region-wide environmental impact assessment. The first steps in developing an effective biosecurity programme are to conduct risk analyses of pathways of introductions, to identify and characterise those species having the highest potential of becoming invasive, and to document impacts to native communities. Major goals of the DoD biosecurity plan are to prevent new introductions and reduce the risk of spread of potentially invasive marine, terrestrial, and freshwater species. The risk analysis process will require the identification of endpoints, hazards, and the likelihood and consequences of different risks.

Plant responses following eradication of goats and rats from Raoul Island, Kermadecs

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Goats (*Capra hircus*) were eradicated from Raoul Island in 1986. Some changes apparent in the vegetation as a consequence were: thickening of the pohutukawa canopy; reduction in the dominance of the invasive aroid lily (*Alocasia brisbanensis*) in the forest understorey; increase in the abundance of a) *Hebe breviracemosa* (from one plant to several discrete natural populations); and b) *Pseudopanax kermadecensis*. Rats (*Rattus norvegicus* and *R. exulans*) were eradicated in 2002, leaving no introduced mammals to affect vegetation. Some plant responses observed following rat eradications are: 100-fold increase in germination of nikau (*Rhopalostylis baueriana*) seeds; *Homalanthus polyandrus* seedlings visible widely on the island; and many orange seedlings (*Citrus sinensis*). Most species that did not fruit in the presence of rats are now fruiting e.g., *Hibiscus tiliaceus*, *Catharanthus roseus*, *Bryophyllum pinnatum* and seedlings of those species are establishing. Consequences of the removal of all mammalian browsing pressure are two-fold. Potentially, vegetation succession can return to natural trajectories. The goals for management of some exotic plant species may need to be revised: ideally, *Catharanthus roseus* and *Bryophyllum pinnatum* should be eradicated.

Management of invasive vertebrate species in the United States

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Many invasive vertebrates have become established in the United States and its territories, including at least 20 mammalian, 97 avian, and 53 reptilian/amphibian species. Species from "100 of the World's Worst Invasive Alien Species" are included in each taxonomic group: domestic cat (*Felis catus*), small Indian mongoose (*Herpestes javanicus*), red fox (*Vulpes vulpes*), goat (*Capra hircus*), pig (*Sus scrofa*), rabbit (*Oryctolagus cuniculus*), rat (*Rattus* spp.), house mouse (*Mus musculus*), grey squirrel (*Sciurus carolinensis*), nutria (*Myocastor coypus*), starling (*Sturnus vulgaris*), Indian common myna (*Acridotheres tristis*), red-vented bulbul (*Pycnonotus cafer*), brown tree snake (*Boiga irregularis*), and red-eared slider (*Trachemys scripta*). I briefly review some of these species and the types of damage they cause. I then review the basic types of methods used for control or eradication of each taxonomic group, including physical, chemical, biological and cultural methods. I discuss some of the challenges in managing these species, including issues with the use of toxicants, land access, public attitudes and monitoring difficulties. Finally, I list some ongoing research and future research needs, including fertility control, improved detection methods, improved attractants, improved barriers, improved capture methods and risk assessment methods.

Damage to plants and seabirds by ship rats *Rattus rattus* on the Ogasawara (Bonin) Islands before eradication

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Damage by ship rats (*Rattus rattus*) to plants and seabirds on the Ogasawara Islands, southern Japan disappeared after eradication campaigns conducted using diphacinone rodenticides. Ship rats damage the twigs of endemic trees, *Ochrosia nakaiana* and *Hibiscus glaber*, and feed on the fruits of *Pandanus boninensis* on Nishijima, a 49 ha uninhabited island. Analyses of the rats' age compositions and food habits suggested that they ate soft tissues of twigs due to the shortage of food in winter. Age compositions of ship rats also showed that the season for plant damage corresponded with that of low breeding activities of the rats and scarcity of preferred foods (January – March). *Pandanus* fruits were found to be gnawed all year round, however, such damage stopped after an eradication campaign in March 2007. In April 2008, we found only 82 *Pandanus* fruits remained undamaged on the island. Ship rats also consumed Bulwer's petrels (*Bulweria bulwerii*) on Higashijima, a 28 ha uninhabited island. The meat and feathers of the seabirds were found in 16 stomachs (36%) of 44 rats caught in traps in June 2008. The average body mass of bird-eating rats was significantly larger than that of non-bird-eaters at the 5% significance level. Bird-eating rats ranged from 167 to 253 g body mass, and they were larger than the Bulwer's petrels (78 – 130 g in general).

Surveillance of mongoose and Amami rabbit by auto cameras during mongoose control programmes on Amami-Ohshima Island, Japan

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An invasive small Indian mongoose (*Herpestes javanicus*) (Family Herpestidae, Order Carnivore, Mammalia) in Amami-Ohshima and Okinawa Islands, and in Kagoshima City in Kyusyu was recognised in 2009 as one of the most damaging invasive mammals in Japan. During 2000-2004 and 2005-2014, some control and eradication programmes against the mongoose were implemented by the Japanese government as a model for conservation of biodiversity in subtropical islands. We used 20-40 sets of auto sensor cameras to monitor mongoose and its impacts on native species, especially on Amami rabbit (*Pentalagus furnessi*), which is an endangered species and one of the flagship species on Amami-Ohshima Island,. Mongooses were recorded in early stage of the operations at rabbit nesting areas. After mongoose control, records of mongoose ceased whereas those for rabbits became more frequent. Even at sites with high mongoose and low rabbit numbers, mongoose records ceased after control and rabbit numbers recovered. These results indicate the vulnerability of the Amami rabbit to mongoose invasion.

Lessons learned from gaining political and community support of Hawai`i's first predator-proof fence at Ka`ena Point Natural Area Reserve

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The coastal strand ecosystem of the Ka`ena Point Natural Area Reserve on the island of Oahu, Hawai`i hosts one of the largest seabird colonies in the main Hawaiian Islands, and contains up to 11 species of endangered plants. It is also one of the most culturally significant sites in Hawai`i where souls are said to leap into the afterlife. Due to the negative impacts of invasive mammals on native species, construction of a predator-proof fence was planned for late 2009 and the five invasive mammal species present will subsequently be removed. Prior to construction, two and a half years of extensive public outreach was completed. These efforts reached over 1800 individuals directly, in addition to the thousands that were reached via 11 printed news stories (both local and national), and airing of seven unique television pieces. As a result of these efforts, what was considered a controversial project has achieved broad public support and resulted in the formation of a community and school group dedicated to helping protect the area. During outreach efforts, extensive ecological monitoring was conducted on both native and non-native species to document the effects of predator removal and to determine how best to eradicate the predators present, with the public occasionally participating in this monitoring. The exclusion and removal of these predatory animals is anticipated to result in an increase in the existing population of nesting seabirds, encourage new seabird species to nest at Ka`ena Point, and enhance regeneration and recruitment of native plants and invertebrates. Perhaps just as significant, this project has increased the public awareness of restoration techniques and will provide the people of Hawai`i with a rare opportunity to visit a restored ecosystem.

AUTHOR INDEX

Author names and paper or abstract titles listed in the order of the first-named author.

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