


special issue:

Feathers to Fur

The ecological transformation of Aotearoa/New Zealand



New Zealand island restoration: seabirds, predators, and the importance of history

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Abstract: New Zealand's offshore and outlying islands have long been a focus of conservation biology as sites of local endemism and as last refuges for many species. During the *c.* 730 years since New Zealand has been settled by people, mammalian predators have invaded many islands and caused local and global extinctions. New Zealand has led international efforts in island restoration. By the late 1980s, translocations of threatened birds to predator-free islands were well under way to safeguard against extinction. Non-native herbivores and predators, such as goats and cats, had been eradicated from some islands. A significant development in island restoration in the mid-1980s was the eradication of rats from small forested islands. This eradication technology has been refined and currently at least 65 islands, including large and remote Campbell (11 216 ha) and Raoul (2938 ha) Islands, have been successfully cleared of rats.

Many of New Zealand's offshore islands, especially those without predatory mammals, are home to large numbers of breeding seabirds. Seabirds influence ecosystem processes on islands by enhancing soil fertility and through soil disturbance by burrowing. Predators, especially rats, alter ecosystem processes and cause population reductions or extinctions of native animals and plants. Islands have been promoted as touchstones of a primaevial New Zealand, but we are now increasingly aware that most islands have been substantially modified since human settlement of New Zealand. Archaeological and palaeoecological investigations, together with the acknowledgement that many islands have been important mahinga kai (sources of food) for Māori, have all led to a better understanding of how people have modified these islands. Restoration technology may have vaulted ahead of our ability to predict the ecosystem consequences of its application on islands. However, research is now being directed to help make better decisions about restoration and management of islands, decisions that take account of island history and key drivers of island ecosystem functioning.

Keywords: behavioural ecology; biological invasion; ecosystem function; eradication; flora; mātauranga Māori; nitrogen; pollen; phosphorus; rat; seabird; shearwater; succession; translocation

Introduction

New Zealand's offshore islands extend from the subtropics to the subantarctic region and are numerous: there are 735 islands >1 ha nationally (Parkes & Murphy 2003). These offshore islands are a major focus for ecological research and conservation biology, in part because of the high endemism of the biota of some of them (Cheeseman 1888; Cockayne 1904; Fleming 1979). However, islands with low levels of endemism are also important because they are, or have until recently been, refugia for species that became extinct after

human settlement of the main islands of New Zealand (Worthy & Holdaway 2002; Innes et al. 2010).

This paper reviews progress in the restoration ecology of New Zealand's offshore islands since 1987. Conservation goals and management on offshore islands have undergone major changes in the last 50 years. The emphasis has expanded from species to ecosystems (Towns et al. 1997), with consideration of the interactions between species, and across terrestrial and marine ecosystems. We identify technological advances in three components of this change: control and eradication of non-native species, population genetics and animal behaviour, and the history and role of disturbance regimes.

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Humans settled and substantially modified many of New Zealand's offshore islands. After settlement of New Zealand by Māori in *c.* 1280 (Wilmshurst et al. 2008), many northern offshore islands were used for gardens because the warm climates and fertile soils were well-suited to growing introduced tropical crops (Davidson 1984). Māori rapidly discovered most of New Zealand's offshore islands, including the remote Chatham Islands (King 1989), and even briefly settled on islands situated in the inhospitable subantarctic region (Anderson 2005). Islands also supplied vital resources for Māori civilisation, such as the widely transported obsidian from Mayor Island (Tuhua), and argillite from Rangitoto ki te Tonga (D'Urville Island; Davidson 1984: island names, latitudes and areas are listed in Appendix 1). Other islands were visited regularly to harvest food, including seabird chicks and seals. In the early stages of European colonisation, islands, including those in the remote subantarctic, were also the focus of intensive hunting for whales and seals (e.g. Musgrave 1866) until this became uneconomic through over-exploitation (Bailey & Sorensen 1962).

Polynesian settlement of New Zealand, as elsewhere in the Pacific, resulted in extinction of 27 bird species (Tennyson & Martinson 2006), especially when the commensal kiore (Pacific rat, *Rattus exulans*) also invaded (Steadman 1995; Worthy & Holdaway 2002). Polynesian clearance of vegetation by fire and earthworks are likely to have much reduced island endemic plant and animal populations. European settlement, which involved introduction of a much larger number of non-native plant and animal species, had even greater effects on many island biotas. As colonisation proceeded, agriculture became widespread on islands, often before road and rail infrastructure was developed on the main islands. Agriculture was attempted on islands ranging from subantarctic Campbell Island (from 1894–1931; McGlone et al. 2007) to the subtropical, volcanically active Raoul Island (intermittently from 1889–1937; West 2002). With developing trade around the New Zealand coast, islands were important points for installation of lighthouses, with attendant settlements. Many islands were almost entirely transformed into landscapes dominated by non-native species, some of which had devastating effects on resident flora and fauna. For example, goats (*Capra hircus*), liberated on Great Island (Manawa Tawhi; Three Kings Islands) as food for castaways, reduced populations of two woody plants endemic to the island, *Pennantia baylisiana* and *Tecomanthe speciosa*, to single individuals (Baylis 1948). The once-widespread flightless wren *Traversia lyalli* survived as a last population on Stephens Island (Takapourewa) until 1895 when it was exterminated by cats (*Felis catus*) brought to the island after development of a lighthouse settlement (Galbreath & Brown 2004). Threats to the endemic biota also came from collectors. In 1902, the last birds of the once-widespread *Mergus australis*, which had survived in a remnant population in the Auckland Islands, were shot for collections (Fleming 1982).

The devastating effects of non-native mammalian predators were all too apparent in New Zealand's early colonial history, and as a result islands were quickly recognised for their value in conservation biology. Resolution Island in Fiordland was established as a nature reserve by the Government in 1891 and Hauturu (Little Barrier Island) and Kapiti Island soon thereafter. These island reserves were established as reserves, principally as sanctuaries for bird species clearly threatened on the main islands. The now-widespread practice of translocating animal species was first put into practice on these islands (Armstrong & McLean 1995), for example, translocating brown kiwi (*Apteryx mantelli*) and great spotted kiwi (*A. haasti*) to Hauturu between *c.* 1903 and *c.* 1919 (Turbott 1961). Recognition of

the conservation potential and the tractability of mammal eradication resulted in an early focus on islands. Early examples of mammal eradication include goats from South-east Island (Rangatira; Chatham Islands) (1916), Kapiti Island (1928) and Great Island (1946), cats from Stephens Island (1925) and pigs (*Sus scrofa*) from Aorangi in the Poor Knights Islands (1936). Benefits included rapid recruitment of the previously very rare tree *Meryta sinclairii* immediately after goat eradication on Great Island (Baylis 1951) and an increase in the population of Buller's shearwaters (*Puffinus bulleri*) on Aorangi from *c.* 100 pairs in 1938 soon after pigs were eradicated to *c.* 200 000 pairs in 1981 (Harper 1983).

Along with the successes, there have also been a number of significant setbacks in New Zealand's island conservation. Soon after attempts to create a refuge for threatened birds on Resolution Island, an invasion by stoats (*Mustela erminea*) in 1908 undid the work of its energetic curator, Richard Henry, resulting in the local extinction of translocated populations of kākāpō (*Strigops habroptilus*) and other species. An invasion by ship rats (*Rattus rattus*) on Taukihepa (Big South Cape Island; southern Stewart Island) in 1964 saw the most recent extinctions of New Zealand-endemic vertebrates, including the last populations of bush wren (*Xenicus longipes*) and greater short-tailed bat (*Mysticina robusta*) (Bell 1978; Towns 2009).

By the mid 1980s, the total area of islands in New Zealand which had never been invaded by any non-native mammals was only 2162 ha (Parkes & Murphy 2003), with only four islands of over 100 ha. Given that such a small area was uninvaded, and considering the extinctions on Taukihepa, the prevention of further extinctions of land birds emerged as a major conservation concern during the 1960s to the 1980s, leading to two areas of activity: translocation and eradication. Numerous translocations of successful translocations of rare land birds such as tieke (saddleback; *Philesturnus carunculatus*) among islands (Lovegrove 1996), and endangered birds from the North and South Islands to smaller islands, were successful safeguard against extinction. Successful examples include the translocation of populations of takahē (*Porphyrio hochstetteri*) and kākāpō from Fiordland to islands as far north as Hauturu (Ballance 2007). Guarding against further extinctions also prompted significant advances in eradicating non-native mammals from islands during the 1960s–80s. Eradications took place on large (> 1900 ha) rugged islands: cats from 3083 ha Hauturu by 1980 (Veitch 2001), goats from 2938 ha Raoul Island (Kermadec Islands) by 1984 (Parkes 1990), and brushtail possums (*Trichosurus vulpecula*) from 1970 ha Kapiti Island by 1986 (Norton 2000). A breakthrough during this period was the first planned eradication of rodents from islands smaller than 20 ha (Thomas & Taylor 2002; Towns & Broome 2003). By 1988, eradications resulted in a cumulative island area of 1721 ha freed of non-native mammals (Fig. 1).

Demonstrable successes in island conservation during the 1960s–1980s coincided with a growing public concern about environmental matters and awareness of the country's flora and fauna (Young 2004). The success of translocations of rare land birds to islands increased the public's desire for greater opportunity to see the results and to participate more actively in the conservation and restoration of islands (Rimmer 2004). By 1987, New Zealand was perceived internationally to be a leader in managing islands for threatened species, with recognition given for notable achievements in island conservation including species translocations, eradication of non-native mammals, and public participation (Simberloff 2002).

Much of the progress in island restoration since 1987 was

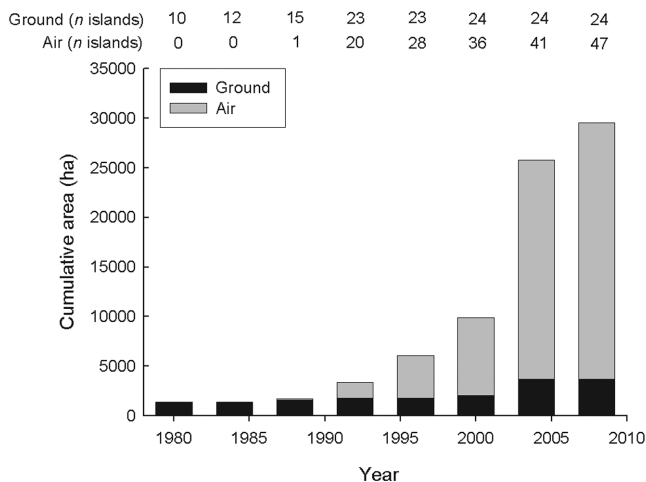


Figure 1. Cumulative area of islands around New Zealand cleared of all non-native mammals since 1980 using ground-based control methods alone (dark bars) or aerial sowing of poison baits (open bars), which for some islands may have been in combination with ground based methods (D.R. Towns, unpublished data). Values shown above the graph are the concurrent cumulative tallies of the number of islands cleared using ground-based and aerial methods.

facilitated by the Conservation Act 1987, which integrated previously competing governmental conservation agencies. Species and habitat protection became the responsibility of the Department of Conservation (DOC). The Department was also given a mandate to advocate for conservation; to seek a broader inclusion of society's views, including those of tangata whenua or first people (Daugherty et al. 1992). We will elaborate on this latter point, since social context has strongly modified progress and will likely do so in the future. However, we begin with eradications of non-native species from islands, because successes are such that New Zealand has developed an "export industry" based on mammal eradications from islands (Simberloff 2002; Rauzon 2007).

Coping with aliens: management of biological invasions

Eradication

By 1987, all non-native mammals except rodents had been eradicated from 13 islands in New Zealand. Ground-based eradication of rodents advanced in the late 1980s, with notable success against Norway rats on Breaksea Island (170 ha) (Taylor & Thomas 1993). However, these methods were not sufficient to achieve eradication of rodents from larger islands with difficult topography. Novel approaches would be needed.

Novel approaches were often informed in part by serendipity. For example, the eradication of Norway rats from Moutohora Island (173 ha) in 1986 was an unintended by-product of attempts to remove rabbits (*Oryctolagus cuniculus*). Carrots dosed with 1080 toxin dropped from aircraft, hand-spread cereal-based Talon 20 P baits containing brodifacoum, wax blocks containing bromodiolone and aerial spread of Talon 20 P by a helicopter with an under-slung bucket (Jansen 1993) eliminated rabbits and Norway rats. Rat eradication was most likely caused by the aerial spread of Talon 20P. Extensive testing of this technique began in

1990 in the Mokohinau Islands using monsoon (fire-fighting) buckets under helicopters (McFadden & Greene 1994). These advances culminated in 1993, with specially designed bait spreaders that could deliver pre-determined bait densities carried by helicopter. When linked with GPS mapping of helicopter flight paths, large scale eradications were finally possible (Towns & Broome 2003).

The formation of DOC in 1987 was instrumental in the success of eradications (Table 1). DOC administered islands where eradication technology could be tested and its researchers worked alongside field staff to develop helicopter bait spread systems with assistance from industry (Towns & Stephens 1997). The newly tested technologies were applied widely, as reported in many examples at a 2001 international conference on eradications (Veitch & Clout 2002). By 2003, the number of non-native mammal species removed from at least one island reached 18, the number of islands cleared of all non-native mammals increased to over 70, and the total area cleared rapidly increased from just under 1500 ha in 1985 (Parkes & Murphy 2003) to over 30 000 ha, which included 65 islands cleared of rats and 12 of mice (Fig. 1). Eradications of rats and rabbits using aerial poison drops have now been achieved at unprecedented scales. The eradication of Norway rats in 2001 from Campbell Island (11 216 ha), surrounded by precipitous cliffs and 700 km from the South Island, was a landmark. Norway rats and kiore were eradicated from Raoul Island (2938 ha) in 2002. Cats were also largely eliminated during this campaign, with the last cat removed in 2004. Kiore were eradicated from Hauturu (3083 ha) in 2004, despite exceedingly steep ravines and cliffs of up to 400 m. By 2008, of the 71 islands from which all non-native mammals have been eradicated, 46 had, at some stage, involved the aerial spread of poison (Fig. 1).

The new technology of aerial poisoning substantially reduced the time required for the successful eradication of non-native mammals. For example, whereas eradication of cats using ground-based operations on Hauturu took place over 3 years (1977–1980; Veitch 2001), cats were eradicated from Mayor Island (1277 ha) simultaneously with rats in one aerial campaign in 2000 (Hunt & Williams 2000).

Eradicating non-native invertebrates and plants on islands remains a challenge. For example, non-native *Vespula* wasps are found on some islands. They are predators of other invertebrates and compete with birds for honeydew (Beggs 2001). Although their control is possible (Harris & Etheridge 2001), there have been no deliberate attempts to remove wasps from islands. However, for unknown reasons, after the eradication of kiore and rabbits from Korapuki Island in 1986–87, *Vespula germanica* disappeared within about five years and has not recolonised (D.R. Towns, unpublished data). An invasion of non-native Argentine ants (*Linepethema humile*) was discovered over about 13 ha of Tiritiri Mātangi Island in 2000, and, beginning in 2001, a deliberate eradication was attempted using fiprinol™ (Harris 2002). By 2008, the ants had been reduced to two small, localized colonies (C. Green, pers. comm.). On many islands administered by DOC, eradications of non-native plant species are conducted according to whether they are of national or local concern. Only on remote Raoul Island is there a systematic approach to eradicating all non-native plant species (West 2002).

Benefits of mammal eradications

The main purpose of eradications of non-native mammals from islands is to protect and enhance native biodiversity. Determination of the extent to which this has been achieved

Table 1. Milestone events in the management of New Zealand islands since 1985

Year	Event	Effect and Key references
1986	First attempt to translocate seabirds: black petrels to Hauturu, 1986–1990	Provided evidence that translocation of birds between burrows would not increase chick mortality (Imber et al. 2003)
1986	First eradication of rats by aerial spread of baits and first simultaneous eradication of non-native mammals (Norway rats and rabbits)	Led to development of systematic methods for aerial sowing of baits (Jansen 1993, Towns & Broome 2003)
1987	Formation of Department of Conservation	Species and site management integrated under a single agency
1988	Address by Atkinson: Opportunities for ecological restoration	Raised the profile and defined opportunities for restoration on New Zealand islands (Atkinson 1988)
1989	Conference on ecological restoration of New Zealand islands	International context for island restoration (Towns et al. 1990a)
1993	First use of purpose-built bait spreaders against rats (Cuvier Island), which completed eradication of all non-native mammals started in 1959	Set scene for systematic eradications of rodents on islands when used in conjunction with GPS (Towns & Stephens 1997)
1994	Completion of planting of 280 000 native trees and shrubs on Tiritiri Mātangi Island	Demonstrated capacity of interested community groups to raise funds and design island revegetation and bird management projects (Rimmer 2004)
1996	First eradication of rats on an islands >1000 ha: Kapiti (1996)	Eradication of more than one species of rat on a large island with challenging topography and tall forest (Empson & Miskelly 1999)
2001	Eradication of Norway rats from Campbell Island (11 300 ha)	Rat eradications in logistically challenging environments and at large scales (McLelland & Tyree 2002)
2005	Recolonisation of Campbell Island by undescribed species of snipe	Rapid recolonisation of oceanic island by an endangered species now able to reclaim entire historic range (Miskelly & Fraser 2006)
2006	Release of captive-bred tuatara to wild on Hauturu two years after eradication of kiore	Population rescued from brink of extinction (only 8 adults known); potential to become largest population
2008	Rat incursion detected and destroyed by trained rodent dog, Motuihe Island	Confirmed capability of rodent detection methods for single invaders
2008	Approval of Ipipiri rat and stoat eradication project, eastern Bay of Islands	Multiple non-native mammal eradication involving collaborations between local authorities, DOC, tourist operators, private landowners, iwi and community groups

is complicated by a lack of baseline data before eradication. This review is therefore confined to those locations for which we could gather information from the literature or from interviews with DOC staff. An assessment of benefits for native biodiversity is possible for 35 islands on which eradications have been completed within the last 20 years (Table 2; see also Appendices 2 and 3). Almost all those eradications were completed since 1990, so responses can be assessed only over limited time. Biodiversity responses on some islands have been spectacular, especially for birds. On Raoul Island, after just six years without rats and with very few cats, five seabird species that had become locally extinct are again breeding on the island (black-winged petrel *Pterodroma nigripennis*; Kermadec petrel *Pterodroma neglecta*; wedge-tailed shearwater *Puffinus pacificus*; sooty terns *Sterna fuscata*; red-tailed tropicbird *Phaeton rubricauda*). Two land birds have recolonised and are found throughout (spotless crane *Porzana tabuensis*; Kermadec parakeet *Cyanoramphus novaezelandiae cyanurus*; K. Baird pers. comm., DRT pers. obs.). All of these birds have almost certainly recolonised from nearby islands in the Kermadec Islands. Within seven years of rat eradication on subantarctic Campbell Island, pipits (*Anthus novaeseelandiae aucklandicus*) spread throughout (Thompson et al. 2005), an undescribed species of snipe (*Coenocorypha* sp.) recolonised (Barker et al. 2005), white-chinned petrels (*Procellaria aequinoctialis*)

and grey-backed storm petrels (*Oceanites nereis*) returned to breed, and an endemic teal species (*Anas aucklandica nesiotis*) was re-established from a population bred on Codfish Island (Whenuahou; P. McClelland pers. comm.). Furthermore, after eradication of kiore from Hauturu, survival of Cook's petrel (*Pterodroma cookii*) chicks to fledging stage immediately changed from a range 5–31% between 1981 to 2003 to 60% in 2005 (Imber et al. 2003a; Rayner et al. 2007).

Benefits for plants and invertebrates have been less well documented. After eradication of kiore from northeastern islands, there was increased seedling recruitment of seven woody plant species, including four canopy trees (Campbell & Atkinson 1999). Similar responses were observed on Breaksea Island in Fiordland after eradication of Norway rats (Allen et al. 1994). On Raoul Island there has been unprecedented seedling recruitment of the endangered endemic shrub *Hebe breviracemosa* after rat eradication. Ten years after eradications of rats and rabbits from Korapuki Island (Mercury Islands), six woody plant species were recruited into the forest understorey and colonized the coast (Towns et al. 1997). Recruitment of ngaio (*Myoporum laetum*) and karo (*Pittosporum crassifolium*) was accompanied by spread of *Coelostomidia zealandica*, a honeydew scale insect (Towns 2002). At least ten other species of large invertebrates also reappeared (Towns et al. 1997). On Raoul Island, a land crab (*Geograpsus grayi*) and

Table 2. Reported benefits to animal species on islands cleared of invasive mammals since 1985, with translocated species marked *, and species with natural recovery on some islands and translocations to others marked (*).

Invertebrates	Flax snail <i>Placostylus ambagiosus</i> , snail <i>Amborhytida taranagensis</i> , ghost crab <i>Ocypode kuhlii</i> , land crab <i>Geograpsus grayi</i> , Cook Strait giant wētā <i>Deinacrida rugosa*</i> , Mahoenui wētā <i>D. mahoenui*</i> , Auckland tree wētā <i>Hemidiena thoracica*</i> , Wellington tree wētā <i>H. crassidens*</i> , Mercury Island tusked wētā <i>Motuweta isolata*</i> , large darkling beetle <i>Mimopeus opaculus*</i> , honeydew scale <i>Coelostomidia zealandica</i> , flax weevil <i>Anagotus fairburni*</i> , Turbott's karo weevil <i>A. turbotti*</i> , knobbed weevil <i>Hadramphus stilbocarpace*</i> , speargrass weevil <i>Lyperobius huttoni*</i>
Amphibians	Hamilton's frog <i>Leiopelma hamiltoni*</i> , Maud Island frog <i>L. pakeka*</i>
Reptiles	Northern tuatara <i>Sphenodon p. punctatus*</i> , Cook Strait tuatara <i>S. p.</i> subsp. unnamed*, Brothers Island tuatara <i>S. guntheri*</i> , common gecko <i>Hoplodactylus maculatus</i> , Duvaucel's gecko <i>H. duvaucelii</i> (*), forest gecko <i>H. granulatus*</i> , Matapia gecko <i>H. sp*</i> , Pacific gecko <i>H. pacificus</i> (*), Marlborough green gecko <i>Naultinus manukanus*</i> , Wellington green gecko <i>N. elegans punctatus*</i> , copper skink <i>Cyclodina aenea</i> , marbled skink <i>C. oliveri*</i> , McGregor's skink <i>C. macgregori</i> (*), Mokohinau skink <i>C. townsi</i> (*), ornate skink <i>C. ornata</i> (*), robust skink <i>C. alani*</i> , Whitaker's skink <i>C. whitakeri*</i> , brown skink <i>Oligosoma zelandicum</i> , chevron skink <i>O. homalonotum</i> , common skink <i>O. nigriplantare polychroma*</i> , Fiordland skink <i>Oligosoma acrinatum</i> , moko skink <i>O. moco</i> , shore skink <i>O. smithi</i> (*), speckled skink <i>O. infrapunctatum*</i> , spotted skink <i>O. lineocellatum</i> (*), Suter's skink <i>O. suteri</i> (*)
Terrestrial birds	Brown kiwi <i>Apteryx mantelli*</i> , little spotted kiwi <i>A. owenii*</i> , brown teal <i>Anas aucklandica chlorotis*</i> , Campbell Island teal <i>A. a. aucklandica*</i> , spotless crane <i>Porzana tabuensis plumbea</i> , North Island weka <i>Gallirallus australis greyi*</i> , New Zealand dotterel <i>Charadrius obscurus</i> , Campbell Island snipe <i>Coenocorypha aucklandica</i> subsp. undescribed, sooty tern <i>Sterna fuscata</i> , kākākāpō <i>Strigops habroptilus*</i> , Kermadec parakeet <i>Cyanoramphus novaeseelandiae cyanurus</i> , red-crowned parakeet <i>C. n. novaeseelandiae*</i> , yellow-crowned parakeet <i>C. auriceps*</i> , kingfisher <i>Halcyon sancta vagans</i> , rock wren <i>Xenicus gilviventris*</i> , pipit Campbell Island <i>Anthus novaeseelandiae aucklandicus</i> , North Island fernbird <i>Bowdleria punctata vealae*</i> , Stewart Island fernbird <i>B. p. stewartiana*</i> , whitehead <i>Mohoua albicilla*</i> , mohua <i>M. ochrocephala*</i> , North Island robin <i>Petroica australis longipes*</i> , South Island robin <i>P. a. australis*</i> , hibi <i>Notiomystis cincta*</i> , North Island tieke <i>Philesturnus carunculatus rufusater*</i> , South Island tieke <i>P. c. carunculatus*</i> , kōkako <i>Calleas cinerea wilsoni*</i>
Seabirds	Grey-faced petrel <i>Pterodroma macroptera gouldi</i> , Kermadec petrel <i>P. n. neglecta</i> , Cook's petrel <i>P. cookii</i> , Pycroft's petrel <i>P. pycrofti</i> (*), fairy prion <i>Pachyptila turtur*</i> , black petrel <i>Procellaria parkinsoni</i> (*), white-chinned petrel <i>P. aequinoctialis</i> , wedge-tailed shearwater <i>Puffinus pacificus</i> , fluttering shearwater <i>P. gavia</i> , little shearwater <i>P. assimilis haurakiensis</i> , grey-backed storm petrel <i>Garrodia nereis</i> , diving petrel <i>Pelecanoides urinatrix</i> , red-tailed tropic bird <i>Phaeton rubricauda reseotincta</i>

a ghost crab (*Ocypode kuhlii*) became apparent or colonised, only after rats were eradicated.

In summary, known beneficiaries of rodent eradications on islands in the last 20 years include 15 species of invertebrates, northern tuatara (*Sphenodon punctatus*), seven species of geckos, 16 species of skinks, 26 species of terrestrial birds and 14 species of seabirds (Table 2). On the 71 islands from which mammals have been eradicated, subsequent responses or translocations have shown potential benefits for 16 species of flightless invertebrates, including critically endangered species such the Mercury Island tusked wētā (*Motuweta isolata*). Among the vertebrates, at least 76 species now have improved prospects, including two of the four species of native frogs; all three taxa of tuatara, 23 of the estimated 80 species of lizards (29%); 32 of the 73 taxa of terrestrial birds (44%), and 16 of the 84 taxa of seabirds (19%). Of this total, at least 57 species have been translocated from other islands or from the mainland. Although less documented, certain species of plants have shown remarkable regeneration when the browsing pressure has been removed (Baylis 1951). As long as there are well-designed measures of change, more biodiversity responses will almost certainly become apparent as additional species recover. However, the outcomes of translocations of many of the more cryptic species, such as invertebrates and reptiles, to islands remain unmeasured, and may remain so for many decades. Furthermore, the gains in indigenous biodiversity that have followed the investment in eradications now need to be defended against reinvasion.

Cost of success: risk of reinvasions

At least 12 of the 18 species of non-native vertebrates on islands were deliberately introduced (Parkes & Murphy 2003). After eradication, only one of these (the stoat) is likely to reach

islands again without human assistance; stoats can swim up to 1.5 km. Of greater concern is possible accidental invasion of islands by four species of rodents that are easily transported by shipping and which are hard to detect, especially given that two of these (ship rats and Norway rats) can swim up to 500 m and thus spread within archipelagos.

There is a danger that the capacity to remove rodents when present at high densities from very large islands has outstripped the capability to detect them at low densities when they first invade. A recent study showed that the rate of incursion (arrival without establishment) by rats can be considerable, with 36 instances of rats arriving on 26 islands over 58 years (Russell et al. 2008a). In just one summer (2007–08), there were three incursions and one near miss (rat detected on a ferry) by Norway rats onto rat-free islands in the Hauraki Gulf. Of more concern were some experimental results: rats carrying radio transmitters avoided detection devices laid out after an incursion, but were detected if the devices were already present when the rats arrived (Russell et al. 2008b). The most unexpected discovery was that a rat wearing a radio collar moved 400m over open sea between Motuhoropapa and Otata islands, despite apparently abundant food resources on Motuhoropapa (Russell et al. 2005).

New techniques using microsatellite DNA provide powerful tools for genotyping populations and locating sources of invading rats. In the Bay of Islands, microsatellite genotyping of ship and Norway rats indicated that Norway rat populations were linked to the adjacent mainland, suggesting dispersal by swimming to and between islands (Miller et al. 2009). In contrast, ship rats could not be traced to the adjacent mainland (where surprisingly they were the only species caught); they probably reached the islands from boats which they boarded outside the immediate area (Miller et al. 2009). The implication

is that the Norway rats on the islands either arrived before ship rats spread into the area, or alternatively, there were episodic irruptions of Norway rats that passed through the ship rat populations on the mainland and dispersed to the islands. With such information, the risks to rat eradication proposals in the Bay of Islands can, for the first time, be assessed using empirical data and can inform population modelling for improved risk assessment (Miller et al. 2009).

Recent innovations have increased the effectiveness of detecting and intercepting invasions, in particular through the use of tracking tunnels with scent lures, and rat-detecting dogs. For example, a single female Norway rat on Motuihe was detected in tracking tunnels, then located and destroyed by a trained dog within 24 hours (F. Buchanan pers. comm.). Perhaps the best criterion for success is the rate of establishment of new rat populations. Outside of the natural swimming range of rats, no new populations have established on islands since 1982, even though the frequency of detected attempts has increased (Russell et al. 2008a). Furthermore, although within the swimming range of rats, Ulva and Limestone Islands have been maintained free of established rats; incursions have been intercepted as they happened (Russell et al. 2008a). The level of surveillance required in the face of regular incursions carries a financial cost, but it is almost certainly far less than the cost of maintaining rats at low densities on the mainland.

Beyond eradications

Island restoration ecology

The ability to eradicate non-native mammals from islands, translocate native animals to them, and defend the islands against reinvasion resulted in a new goal: the restoration of island ecosystems. Ecological restoration through reconstruction of plant communities has been conducted through planting trees on several deforested New Zealand islands since the 1920s. On Kapiti Island, volunteers from the Royal Forest and Bird Protection Society planted trees, including non-native species, on former farmland between 1925 and 1943 to create habitat for land birds (Esler 1967). Revegetation using native tree species was attempted in the 1970s on Mangere Island (Chatham Islands) to provide habitat for translocated black robins (Atkinson 1988). The largest and most successful project involving revegetation of a deforested island was on Tiritiri Mātangi Island, where 280 000 locally-derived seedlings of woody plants were planted between 1984 and 1994 (Rimmer 2004).

From the late 1980s, restoration ecology benefited from newly emphasised links with applied ecology (Jordan et al. 1987) and recognition of the importance of population viability studies and genetics to conservation biology (Simberloff 1988). Atkinson (1988) outlined the potential for ecological restoration to assist with biodiversity declines in New Zealand and the particular role that islands could play. Ecological restoration in this context was explicitly interventionist. It required definition of goals for the composition of restored communities and Atkinson emphasised the need to monitor progress towards those goals.

Atkinson's (1988) paper had such impact that ecological restoration became the theme of an island management conference in 1989 (Townes et al. 1990a). The conference coincided with new opportunities for island management provided by the Conservation Act, including protection interpreted in the context of enhancement (Townes et al. 1990b). At the conference, Atkinson (1990) advocated a systematic approach to island management encompassing

a spectrum of management goals that could accommodate the range of actions, from minimal intervention on some islands to high levels of intervention on others. An address by Simberloff (1990a) outlined the conceptual challenges in defining goals for restoration of islands. He likened restoration to a fuzzy target because of uncertainties about the nature or development of communities and ecosystems. Furthermore, because of the dynamism of natural communities, the target was identified as mobile, and therefore unable to be seen or characterized. Simberloff (1990b) also emphasised legacy effects and potentially irreversible ecological consequences of biological invasions.

Translocations: beyond euphoria

Species translocations remain an essential element of ecological restoration of some islands. Translocations are also conducted to provide sanctuary for species on islands where the species may never have occurred. Despite a particularly high success rate for translocations in New Zealand (Armstrong & McLean 1995), not all succeed. Hihi (*Notiomystis cincta*) was widespread on the North Island until 1872, but became extinct throughout its range except for one remnant island population on Hauturu (Oliver 1955). Translocations of hihi derived from that population to other islands failed consistently (Castro et al. 1994). One postulated reason was that forests had not developed sufficiently to provide year-round sources of nectar, but this was not supported by comparative analyses of supplementary feeding, time budgets and body condition of the birds (Armstrong & Perrott 2000). By 2007 the only translocated hihi populations still surviving were those under intensive management using some artificial sources of nectar (Armstrong et al. 2007).

The success of translocations of land birds can usually be assessed within a few years of the attempt. But for many other species, the results of translocations are unknown and may remain so for decades. For example, of 26 reptile species translocated to islands, only four individual attempts have so far passed the most basic criterion for success by demonstrating that the populations are self sustaining (Townes & Ferreira 2001, D.R. Townes unpublished data). The problem is the low annual reproductive output of many species, which for some, such as geckos, may be 1–2 young per female per annum (Cree 1994). With the added difficulties of cryptic behaviour and attendant sampling problems, some lizard species may require 20 years or more of monitoring to determine the success of their translocation (Townes & Ferreira 2001). Tuatara are even more problematic because low annual reproductive output is coupled with slow maturation: tuatara take at least nine years to reach reproductive age (Castanet et al. 1988).

Even if the translocations succeed, there are likely genetic legacies of massive population declines before translocation or of the repeated sampling of constrained populations. For example, the present Chatham Islands population of about 250 black robins (*Petroica traversi*) originated from five birds (Ballance 2007), including a single breeding pair (Butler & Merton 1992). The survival of this population and success of other translocations of birds indicate that, under the right conditions, small founder populations with two males and two females of birds can succeed (Taylor et al. 2005), at least in the short term. However, minisatellite DNA data have revealed that black robins have among the lowest genetic variation reported for any wild bird species (Adern & Lambert 1997).

The effects of inbreeding depression on such populations have largely been ignored, with suggestions that any detrimental

effects are less than those seen in continental regions (cf. Jamieson et al. 2006). However, genetic issues can arise in small populations. Small populations of takahē translocated to islands are of inbred stock and consistently show elevated egg infertility and fewer “juveniles per egg” compared with the source population in Fiordland (Jamieson & Ryan 2000). For takahē, inbreeding appears to have reduced reproductive fitness only in females (Jamieson et al. 2003). Kākāpō conservation face similar problems. The entire population of kākāpō has been translocated to islands. These birds have low egg-hatching success, with only 42% hatching success between 1995 and 2002 (Elliott et al. 2006). Unusually high frequencies of sperm deformity (D. Eason pers. comm.) indicate that loss of fitness resulting in reproductive failure may be expressed through male kākāpō.

Inbreeding depression can also result in increased susceptibility to disease. Disease resistance in vertebrates is thought to be mediated through variation of gene loci in the major histocompatibility complex (MHC) (Jamieson et al. 2008). Black robins have low variation in MHC genes compared with South Island robins (*Petroica australis australis*), but the long term effects of low variation remains unclear (Miller & Lambert 2004).

Although there may be no genetic link, success of at least two translocations was compromised by outbreaks of disease in the translocated birds. In 2004, three female kākāpō died soon after their transfer to Chalky Island. All three had contracted a bacterium causing erysipelas from contact with soil and seabirds on Codfish Island before departure (Ballance 2007). Similarly, high mortality of translocated hihi on Mokoia Island was linked to outbreaks of nest mites (Armstrong et al. 2007) and aspergillosis, with the latter associated with stress in the highly managed bird population (Alley et al. 1999). The fungus *Aspergillus fumigatus* inhabits soil and can be carried on soil-inhabiting invertebrates eaten by hihi (Alley et al. 1999). The difficulties of managing this fungal threat proved insurmountable and all hihi were eventually removed from the island (Armstrong et al. 2007).

The effects of small populations on translocation success are not confined to birds. All six island translocations of Mercury Island tusked wētā are from populations raised in captivity derived from one male and the more prolific of two females (Stringer & Chappell 2008, I. Stringer pers. comm.). The potential to compensate for any long term effects of this narrow genetic base on the translocated populations may be limited by the tiny source population on Atiu (Middle Island; Stringer & Chappell 2008). Microsatellite DNA analysis of all tuatara populations indicates that the *Sphenodon guntheri* population confined to North Brothers Island has very low genetic variation, possibly due to a small population founder effect. Inbreeding effects may have contributed to the unusually high (60%) male bias, low reproductive output (9% gravidity per annum), and declining body condition (MacAvoy et al. 2007).

Behavioural impediments and breakthroughs

Studies of animal behaviour have been an important component of conservation biology on New Zealand islands (Moore et al. 2008), despite doubts that theoretical advances in animal behaviour would find utility in conservation (Caro 2007). Indeed, management of populations of several animals on islands was hampered until behavioural issues were resolved.

Advances in behavioural understanding that have been pivotal in the conservation of island birds include the cross

fostering of black robin eggs into nests of Chatham Island tits (*Petroica macrocephala chathamensis*) to increase productivity (Butler & Merton 1992) and the discovery of lek breeding behaviour in kākāpō (Merton et al. 1984). Lek breeding behaviour (which is unusual for a parrot) gave clues to when breeding was likely as leks (track and bowl systems) were modified by the birds and their booming calls heralded likely mating (Ballance 2007). However, even in booming years, productivity remained low and most chicks produced were male. The use of supplementary feeding in an attempt to increase the number of kākāpō chicks had limited success (Elliott et al. 2006). Furthermore, comparative analyses showed that those females that had eaten supplementary food produced an excess of male offspring, an outcome predicted by sex allocation and parental investment theory (Clout et al. 2002). A new, optimized feeding regime was then instigated and female weights were raised only in birds predicted to be below the threshold for breeding. By limiting feeding to keep females just above this weight, male-biased offspring sex ratios were avoided (Robertson et al. 2006).

Behavioural studies of social interactions of birds have also helped to inform translocations to islands, either to retain the new birds on site or to attract others to colonise. For example, an understanding of the cooperative breeding behaviour of whiteheads (McLean & Gill 1988) was fundamental to their successful translocation to Tiritiri Mātangi Island (Rimmer 2004). After eradicating predators from some islands on which seabirds were likely to have been present in the past, natural recolonisation has been extremely slow or absent. For example, grey-faced petrels (*Pterodroma macroptera gouldi*) have not recolonised Hauturu, despite eradication of their main predator (cats) in 1980. This is not surprising since many species of seabirds show high philopatry; if transferred as adults, most seabirds return to their island of origin (Imber et al. 2003b). In a study beginning in 1986, 249 black petrel (*Procellaria parkinsoni*) chicks close to fledging were transferred from the large breeding population on Great Barrier Island (Aotea) to supplement a population previously depleted by cats on Hauturu (Imber et al. 2003b). Over almost 10 years, only 11 transferred birds were found, and nine of these had returned to Great Barrier Island rather than Hauturu. A similar study translocated 334 fluttering shearwater (*Puffinus gavia*) chicks from Long Island to Maud Island in 1991, where they were artificially housed in burrows and hand-reared until fledging (Bell et al. 2005). By 2004, 32 of the 273 chicks that fledged (12%) had returned to the island, which now supports 15 breeding pairs. Similar methods were employed to encourage the establishment of fluttering shearwaters, common diving petrels (*Pelicanoides urinatrix*) and fairy prions (*Pachyptila turtur*) on Mana Island (Miskelly et al. 2004). As an additional attractant, solar-powered sound systems were used to broadcast calls of the birds during hours of darkness. Of 239 diving petrel chicks transferred and hand fed on the island during 1997–99, about 50% fledged and 20 returned to the island. In addition to these, however, 51 birds colonised without assistance near the sound system, and these were the first birds to breed on the island. By 2003, 42 adult diving petrels were resident on Mana Island, 19 of which had formed pairs (Miskelly & Taylor 2004).

Successes with eradications, invasion detection and translocations have laid the ground for ecological restoration. The plantings on Kapiti and Tiritiri Mātangi Islands have also shown what is possible when revegetating deforested sites. However, without clear goals for restoration, well-intentioned projects could suffer from conflict between special-interest

groups that advocate for particular species (Atkinson 1990). If we are to move from well-intentioned to well-planned, a great deal more needs to be known about the history and biogeography of restoration sites themselves.

From ecological reconnaissance to syntheses

Comprehensive descriptions of the flora and fauna are fundamental to mechanistic understandings of the ecology and evolution of island biotas. During the last thirty years there has been little institutional support for the production of multi-disciplinary monographs on the physical environment and biota of islands as in the past (e.g. Whakaari or White Island, Hamilton & Baumgart 1959; Hauturu, Hamilton 1961). There has been a reduction in commitment of universities to maintaining remote research facilities on islands which in the past resulted in series of published papers (e.g. University of Canterbury's Snares Islands expedition papers, including Warham and Wilson 1982). Nonetheless, there has been some progress towards improved fundamental knowledge of islands during the last 20 years. New taxa have been found or described from islands, such as an undescribed snipe on Jacquemart Island (Miskelly & Fraser 2006) and the shrub *Pimelea telura* from the Three Kings Islands (Burrows 2008). Monographs on the biota of islands with high levels of endemism have been published (e.g. flora of Aorangi in the Poor Knights Islands; de Lange & Cameron 1999). Descriptions of plant communities have been produced for other islands, along with maps of their distributions (Stewart Island, Wilson 1987; Campbell Island, Meurk et al. 1994a; The Snares, Hay et al. 2004; Rangitoto, Haines et al. 2007). Yet despite repeated visits to many islands, basic descriptions are, in many cases, still lacking.

We do not mean to imply that there is a dearth of information about New Zealand's islands. Comprehensive records about the biota of many islands have been used in recent meta-analyses of determinants of successful invasions by non-native mammals. An analysis of 164 introductions of six non-native mammals to 103 islands throughout New Zealand revealed that if a population persisted longer than 25 years on an island it was unlikely to become extinct (Duncan & Forsyth 2006). The likelihood of extinction of an introduced population on an island was greater the higher its latitude. Data on the presence of rodents from 297 islands throughout New Zealand showed that the current distribution of kiore is strongly negatively related to the presence of ship rats and to a lesser extent to the presence of Norway rats (Russell & Clout 2004). Non-native mammal species richness across the same islands was principally a function of human introduction effort and continued human use (e.g. agriculture), and distance from the North and South Islands as the main sources was important for mammals <10 kg body weight (Russell et al. 2004). These examples show the value of the collection of consistent data. Indeed, information about non-native vertebrates from New Zealand's islands contributes disproportionately to many global reviews and meta-analyses (e.g. Moulton & Sanderson 1997; Courchamp et al. 2003; Jones et al. 2008).

Despite the dynamic nature of the biota of many New Zealand islands, repeat surveys of the biota of islands are seldom undertaken. Some exceptions include surveys of the floras of Atiu or Middle Island, Mercury Islands (Atkinson 1964; Cameron 1990) and Aorangi, Poor Knights Islands (Cockayne 1905; de Lange & Cameron 1999), and surveys of

birds on the Mokohinau Islands (Sandager 1890; McCallum 1980). Analyses of repeated censuses of islands' biotas could provide insight into assembly rules on islands, the role of dispersal and the responses of biota to climate change.

Islands as dynamic systems

Ecologists' changing perceptions

The origins of New Zealand's extant native biota remain a subject of intense research interest (Gibbs 2006; Tennyson 2009). Some islands, such as the Chatham Islands, have a diverse endemic flora and fauna but are comparatively young (1–3 Ma; Trewick et al. 2007) and their biota has most likely originated through long-distance dispersal. An implication is that species pools on islands are dynamic over time. Examples of new dispersal and establishment by native biota on remote islands within the last 20 years include nau or Cook's scurvy grass (*Lepidium oleraceum*) to the Bounty Islands (Amey et al. 2007) and welcome swallow (*Hirundo tahitica*) to the Chatham Islands (Freeman 1994). Such events are indicative of the importance of considering changing species pools on islands over ecological time scales as well as over evolutionary time scales (McGlone 2006).

Because of high levels of endemism and the abundance of once-widespread species such as tuatara, early ecologists in New Zealand described some islands as "pristine" and "primaeval" (e.g. Cochrane 1962). We argue that these views, imbued in part by romanticism, have been counterproductive for understanding the ecology of islands and for setting realistic goals for their restoration. There is growing realism about the dynamism of island ecosystems and in particular about the role of human activity in determining the current state of island ecosystems. For example, the first account of the vegetation of the Aldermen (Ruamāhua) Islands, east of the Coromandel Peninsula, in 1925 (Sladden & Falla 1927), did not recognise that the "natural meadow" cover present on the islands at that time was fire-induced. An updated account of the vegetation in 1972 recognised that the forest, which by then covered much of the islands, had developed after cessation of repeated fires used by Māori since at least 1830 to facilitate access to petrel burrows for harvesting of chicks (Court et al. 1981). For many islands in New Zealand, the role of past fire management by Māori is now generally acknowledged as important in understanding past and contemporary vegetation (Hawke et al. 2003; Atkinson 2004).

We also contend that regarding islands free of non-native mammals as primaeval selectively ignores influences from other non-native biota. Little attention has been paid to the role of many non-native species (notably plants, birds, and invertebrates) on offshore island ecosystems; this is in contrast to the main islands of New Zealand where effects of non-native biota have been a major area of research. Non-native plant species have colonised nearly all of New Zealand's offshore islands. Some of these species are controlled on Nature Reserves such as Raoul Island (Sykes & West 1996), but on many islands other species appear to be ignored either because of untested assumptions that their effects (above- and below-ground) are negligible or because of a tacit acceptance that their control or elimination is too difficult or potentially too damaging to undertake. For example, the bird-dispersed non-native herb *Phytolacca octandra* is a widespread component of vegetation on most rodent-free, warm temperate islands and a plant that rapidly colonises canopy openings (Cameron 1990), yet it

is seldom subject to control and it is unknown whether its colonisation of canopy openings affects longer term community composition during forest development in these sites.

Non-native land bird species have colonised even the most remote islands (Heather & Robertson 1996) and are not subject to control on any islands. These species are expected to dominate bird communities in early successional plant communities on islands, but not in old-growth forests (Diamond & Veitch 1981); an expectation supported by empirical evidence from Hauturu (Kikkawa 1964; McCallum 1982). However natural and human disturbance has resulted in early successional communities dominating most of New Zealand's islands (e.g. Sykes 1977; Clarkson & Clarkson 1994; Atkinson 2004), so competitive interactions between non-native and native land birds can be expected. For example, non-native seed-eaters such as redpolls (*Carduelis flammea*) co-occur with various island-endemic seed-eating subspecies of kākārīki (*Cyanoramphus novaezelandiae*), but it is unknown whether they compete for food. Non-native birds are also likely to determine ecosystem processes on islands. Starlings (*Sturnus vulgaris*) breed from the Kermadec Islands to Campbell Island and disperse seeds of native species (Baylis 1986; Ferguson & Drake 1999) and non-native species (Cameron 1990, Anderson et al. 2006) within islands and from the mainland to islands. Indeed there may be greater likelihood of seed dispersal by starlings to rat-free islands because starlings roost preferentially on them (Atkinson 1964; Brockie 1983).

Non-native invertebrates have been shown to have profound ecosystem impacts on some islands. Non-native crazy ants (*Anoplolepis gracilipes*) on Christmas Island (Indian Ocean) prey upon native land crabs, the dominant endemic consumer on the forest floor, which in turn alters forest regeneration patterns (O'Dowd et al. 2003). These ants have increased to form a large part of the island's animal biomass, facilitated by positive interactions with native honeydew-producing scale insects (Abbott & Green 2007). Similarly, in New Zealand *Nothofagus* forests, food webs based on honeydew scale insects (*Ultracoelostoma* spp.) have been altered by non-native *Vespa* wasps (Beggs & Wardle 2006). Honeydew-based food webs are also found in forests on islands in New Zealand (e.g. Towns 2002), but it is unknown whether non-native invertebrates alter these food webs.

A more comprehensive view of island ecosystems, in terms of both native and non-native components, has resulted in more cautious views of effects of eradications (Zavaleta 2002) and translocations to achieve restoration. For example, there is concern that non-native molluscs on Mahuki Island (most likely *Deroceras reticulatum*; J. Boow, pers. comm.) could be released from current predator pressure from ship rats and thus threaten the current large population of threatened Cook's scurvy grass. Further investigation could be needed before the planned eradication of rats takes place. Plans for restoration of ecosystem processes in the Kermadec Islands could include the reintroduction of the New Zealand pigeon (*Hemiphaga novaeseelandiae*) which became extinct in the Kermadec Islands in the 1800s (Worthy & Brassey 2000), because it plays a key role in the dispersal of large-seeded native trees (Clout & Hay 1989). However, there are concerns that if pigeon populations were re-established, they could also disperse non-native trees such as guava (*Psidium guajava* and *P. littorale*) which are currently under control (D. Havell, pers. comm.).

Palaeoecological perspectives

An increasing body of long-term fossil (especially pollen) evidence from islands informs a more dynamic view of the biota of islands, including extinctions and colonisations. Palaeoecological studies of pollen cores from the Chatham Islands (Erdtman 1924) were amongst the first conducted in New Zealand. Reconstruction of the Holocene vegetation history has been developed from pollen in cores in the peaty soils on the sub-Antarctic Islands (Moar 1958, 1959; McGlone et al. 2000), Stewart Island (Rakiura; McGlone & Wilson 1996) and the Chatham Islands (Mildenhall 1976). For example, on Auckland Island after glacial retreat there was initial dominance by tundra communities, followed by shrub and grassland until 5500–4000 ¹⁴C yr BP, after which a forest cover persisted, dominated by southern rātā (*Metrosideros umbellata*) (McGlone et al. 2000). Recent palaeoecological studies are also providing insight into past human influences on islands. On subantarctic Campbell Island, pollen evidence suggests that the recent spread of woody vegetation (especially *Dracophyllum* spp.) is principally due to secondary succession after the abandonment of sheep farming and burning on the island (activities that occurred between 1894 and 1931), rather than being attributable to climate change (Wilmshurst et al. 2004; McGlone et al. 2007). Pollen records from swamps on Great Barrier Island (Horrocks et al. 2001) and Mayor Island (Empson et al. 2002) showed Māori settlement impacts on two islands in northern New Zealand, where conifer-rich forests were cleared by fire (c. 600 and c. 450 ¹⁴C yrs BP respectively) and replaced with fire-induced communities dominated by bracken (*Pteridium esculentum*). In addition, studies of pollen records and preserved rat-gnawed seeds excavated from peaty deposits on the New Zealand mainland (Wilmshurst & Higham 2004; Wilmshurst et al. 2008), and rat-predation on terrestrial landsnail shells in Northland (Brook 2000) demonstrate a variety of approaches that can help pinpoint the timing of initial rat arrival on islands and track the potential impact of predation on the biota through time (Prebble & Wilmshurst 2009). Records such as these can provide information to guide goals for restoration of islands by providing information about whether plants suggested for introduction were present formerly, the timing and nature of previous extinctions and introductions, and the impact of rat predation. Fossil vertebrates also provide insight into past human impacts on islands. On the Chatham Islands, middens and other deposits provide evidence of a diverse community of land birds and seabirds, many of which did not survive human settlement or, like taiko (*Pterodroma magentae*), barely survived it (Lawrence et al. 2008).

A more comprehensive view of the role of human impacts on the patterns and processes on islands is likely to emerge from ecologists working with a broader range of disciplines, for example with archaeologists, historians and the Māori owners or kaitiaki (guardians) of islands about which they have centuries of mātauranga (traditional knowledge). As an example of how mātauranga can inform and enhance ecological investigations, the rate at which sooty shearwater (tītī, *Puffinus griseus*) chicks were harvested for food from Poutama Island (Foveaux Strait) in a given year relative to the previous year's harvest over a 19 year period was a strong predictor of the onset and strength of an ENSO event in the following 12 months (Lyver et al. 1999).

The crucial role of seabirds in island ecosystems

Seabirds: a key component of biodiversity and ecosystem function

The focus on conservation biology on New Zealand's islands during most of the 20th century was on its land biota, in particular land birds and reptiles. It is rather surprising that seabirds have received less attention because they are a significant component of New Zealand's biodiversity, especially on islands. Of a total world seabird fauna of 350 species, 140 (39%) have been recorded from New Zealand. With 84 species (23% of the world's total fauna) breeding in New Zealand – the vast majority of these confined to offshore islands – New Zealand is the world's "seabird capital" and 35 species (10%) breed nowhere else (Taylor 2000). Feeding ecology and demography of some of the breeding seabirds have been studied on islands throughout New Zealand (e.g. Imber 1973; Walker & Elliott 1999). Marine resources have been clearly demonstrated to be the principal driver of population trends in seabird populations (e.g. Lyver et al. 1999; Jackson et al. 2005; Shaffer et al. 2006; Mills et al. 2008).

Seabirds play a crucial role in island ecosystems as providers of marine nutrient subsidies (Gillham 1956a, 1960a; Atkinson 1964) and as a major part of islands' natural disturbance regimes (Gillham 1956b, 1960b, c). Despite this being well established in these pioneering studies, the emphasis on land biota of islands in conservation biology between the 1960s and 1980s set agendas for later island restoration proposals, with little, if any, emphasis on the role of seabirds (e.g. Miller et al. 1994; Rimmer 2004). A recent renewal of interest in the subject of seabird effects on New Zealand ecosystems stems from better appreciation of their once-widespread role. Some seabird species suffered range declines or became extinct after Māori settlement. For example Scarlett's shearwater (*Puffinus spelaesus*) did not survive beyond c. 600 yr BP (Worthy & Holdaway 2002) and others were eliminated from the main islands of New Zealand. Many species of shearwaters and petrels remained widespread until soon after European settlement, not only in coastal regions but breeding on mountains as far inland as the Kaimanawa Ranges (Oliver 1955). Besides fossil evidence, there is also the legacy of marine nutrients in soils from colonies extirpated after European colonisation (Hawke et al. 1999). A few shearwaters and petrels are now confined to the main islands of New Zealand including Westland petrel (*Procellaria westlandica*) and Hutton's shearwater (*Puffinus huttoni*). For most of New Zealand, the predators introduced by Europeans eliminated most formerly widespread populations. Most shearwater and petrel species now breed only on islands, sometimes in very high densities (McKechnie 2006), although a few species (e.g. sooty shearwater) also maintain small populations on coastal headlands of the main islands.

A desire to restore island ecosystems has been instrumental in putting the focus in ecological research back onto seabirds and their effects on ecosystems (Miskelly 1998; Towns & Atkinson 2004). Seen in these terms, seabirds are ecosystem drivers and much of the land fauna merely passengers (cf. MacDougall & Turkington 2005). Burrowing fairy prions added 4–50 g m⁻² of guano to the ground surface per week on Stephens Island, enriching soil phosphorus (P) and nitrogen (N) concentrations (Mulder & Keall 2001). High soil N from marine sources brought to the land by seabirds resulted in higher foliar N concentrations in plants around seabird colonies

in southern New Zealand than in nutrient-poor areas without seabirds (Hawke and Newman 2007). However, more intensive burrowing by seabirds was associated with lower soil pH (Mulder & Keall 2001; Roberts et al. 2007), and reduction in both the emergence of seedlings from the soil seed bank and the growth of phytometer plants (Mulder & Keall 2001). The high N and P concentrations and high acidity resulting from input of large amounts of guano can still be beneficial for some species, notably endemic herbs. For example, Cook's scurvy grass was abundant when encountered by early European navigators, but its range has contracted drastically, with most populations now on islands where they are associated with guano in colonies of gannets (*Morus serrator*), gulls and shags (Norton et al. 1997). Although seabirds are key drivers of soil nutrient status and disturbance regime on many islands, there are also other important direct influences of the marine environment, especially in windy sites where marine aerosols influence soil chemistry (Meurk et al. 1994b) and salt spray affects vegetation structure (Gillham 1960d).

Introduced predators and their effects on seabird-dominated islands

Recent research has contrasted ecosystem processes on islands dominated by burrowing seabirds with islands where ship or Norway rats have exterminated or severely reduced seabirds. Soils on rat-invaded islands in northern New Zealand had 47% less total carbon (C), 45% less total N and 53% less total P than soils on uninvaded islands, and marine-derived $\delta^{15}\text{N}$ in soils was 23% lower (Fukami et al. 2006). Reduced soil resources on rat-invaded islands has resulted in reduced microbial CO₂ production and reduced densities of soil organisms, notably herbivorous and microbe-feeding nematodes, collembola and minute land snails (Fukami et al. 2006; Towns et al. 2009). There is also evidence that following extirpation of seabird colonies by rats on these islands, the forests on them have become increasingly N- (although not P-) limited (Wardle et al. 2009). Foliar and litter N concentrations across a range of widespread plant species differed between uninvaded and invaded islands and in all cases they were lower on rat-invaded islands. Additionally the ratio of N to lignin in litter on these islands was greater and litter decomposition rates were slower (Wardle et al. 2009). On uninvaded islands, with abundant burrowing seabirds, just over half the C stored is in non-living pools below ground (Wardle et al. 2007). In contrast, on rat-invaded islands most of the C is stored in live plants and the total amount of C stored exceeded that on uninvaded islands by 36.7% (Wardle et al. 2007). The most likely reason is that rat invasions ended the intense burrowing and trampling disturbance caused by seabirds. On densely seabird-burrowed Rangatira, exclusion of seabirds for 3 years increased tree seedling densities 13-fold (Roberts et al. 2007). The extermination of seabirds by rats achieves the same effect, thus rat-invaded islands have greater seedling densities and enhanced tree basal area (Fukami et al. 2006).

The increased density of seedlings when rats are present and seabirds are absent or rare (Fukami et al. 2006) disguises the impacts that rats can have on seedling densities of individual tree species. Selective seed predation by kiore could alter long-term forest composition (Campbell & Atkinson 1999, 2002). For example, the tree *Streblus banksii* is dominant on some islands never invaded by rats (e.g. Atiu or Middle Island, Cameron 1990) and its seedling recruitment is substantially reduced on islands where kiore are present (Campbell & Atkinson 2002). However experiments are needed to determine

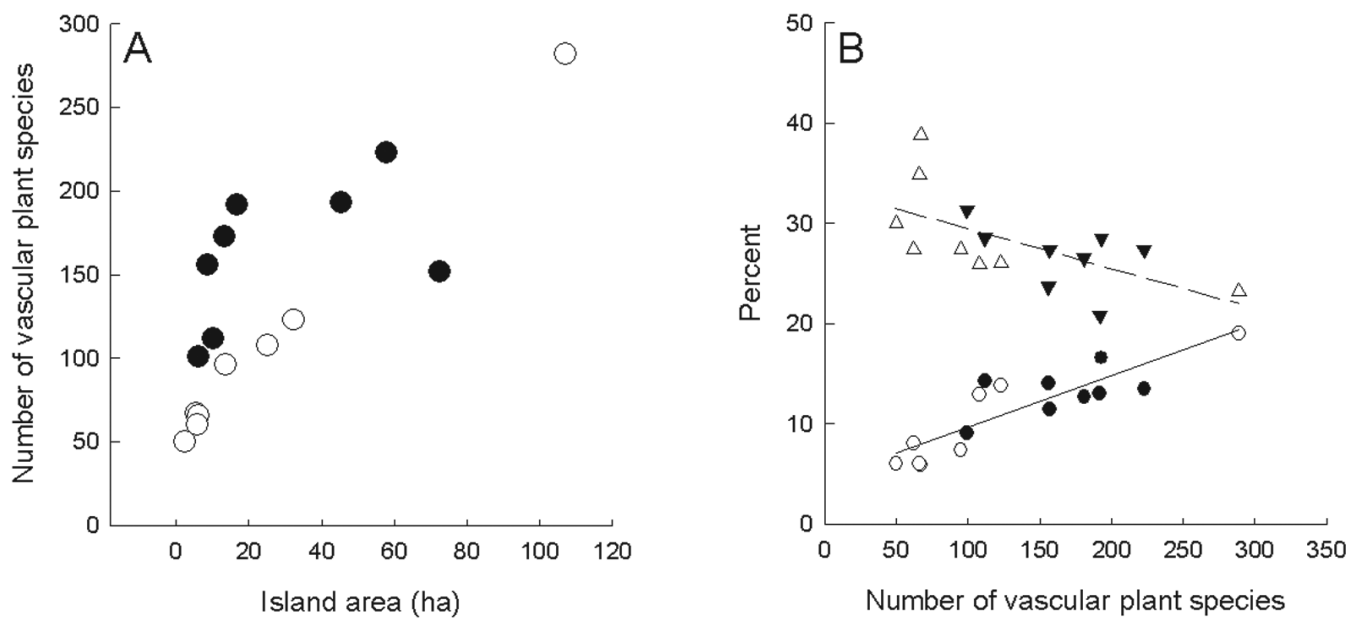


Figure 2. A. Relationship between island area and the number of vascular plant species on 16 islands in warm temperate northern New Zealand; open symbols = islands never invaded by rats; filled symbols = islands invaded by ship or Norway rats. B. Relationships between the number of vascular plant species on an island and the percentage of species comprised by woody plants including lianes (triangles and scored line: $R^2 = 0.340$ $P = 0.010$), or ferns (circles and solid line: $R^2 = 0.710$ $P < 0.001$). Open symbols = islands never invaded by rats; filled symbols = islands invaded by ship or Norway rats. Herbs (not shown) showed no relationship; see text. Data sources listed in Appendix 4.

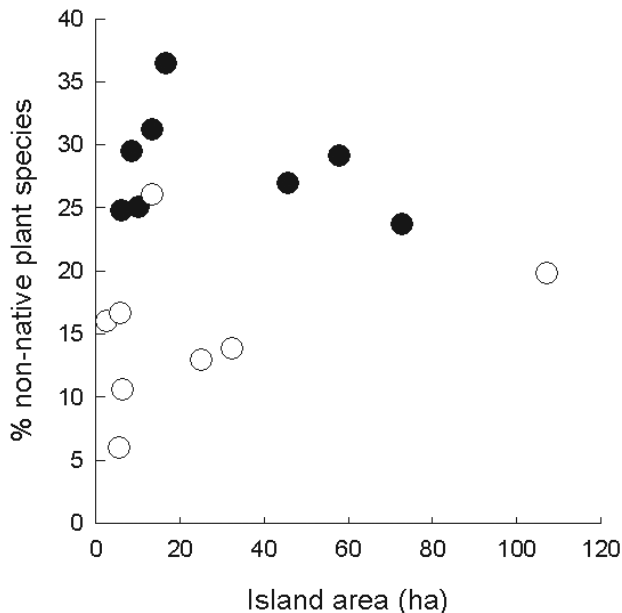


Figure 3. Relationship between island area and the percentage of vascular plant species that are non-native on 16 islands in warm temperate northern New Zealand; open symbols = islands never invaded by rats; filled symbols = islands invaded by ship or Norway rats. Data sources listed in Appendix 4.

whether its dominance on islands with abundant burrowing seabirds is because of lack of seed predation or because it has greater tolerance of seabird disturbance than other co-occurring tree species.

To evaluate effects of seabirds and their predators (rats) at a whole-island scale, we compared the composition of the floras of warm temperate northern New Zealand islands. All are forested islands with vegetation typical of those in the region, i.e., vegetation that has developed after fire and in some cases permanent Māori settlement and agriculture, now abandoned (Atkinson 2004). Eight of the islands have never been invaded by rats and have large populations of burrowing seabirds (mean size 25 ha; range 2.5–107.1 ha), while the other eight islands have been invaded by ship or Norway rats and have few or no burrowing seabirds (mean size 29 ha; range 6.2–72.7 ha); these were a subset of the islands studied in some recent research (Fukami et al. 2006; Wardle et al. 2007; Mulder et al. 2009). The expected relationship between island size and number of plant species was found across the 16 islands (Fig. 2a). As plant species richness on islands increased, so did the percentage of the flora that was ferns (Fig. 2b). There was a concurrent decrease in the percentage of the flora comprised of woody plants and lianes, 96% of which were dicotyledons. Woody monocotyledons, especially *Cordyline* spp., comprised most of the remainder, while conifers, both native and non-native, were on average 1% of the flora and were absent from 11 of 16 islands. There was no significant relationship between the species richness of island floras and the percentage of the flora that was herbs ($R^2 = 0.043$ $P = 0.44$; also, there were no relationships when dicot and monocot herbs were considered separately).

In addition to island size (Fig. 2a), invasion status was a determinant of the total number of vascular plant species, which was greater on islands invaded by rats (163 ± 14.7 [SEM] species per island) than on uninvaded islands (107 ± 27 species

per island; ANCOVA: Island invasion status $F_{1,13} = 8.47$, $P = 0.012$, covariate island size $F_{1,13} = 32.2$, $P < 0.001$). Of 531 vascular plant taxa across the 16 islands, 199 were not found on any of the uninvaded islands and 103 were not found on any of the invaded islands (comparing all uninvaded vs. all invaded islands' floras: Jaccard's similarity index = 0.43; Sørensen's qualitative index = 0.60; relative scores on a scale between 0 = no similarity and 1 = complete similarity). The percentage of vascular plant species that were non-native was not a function of island size, but was strongly influenced by island invasion status (rat-invaded islands, $28.3 \pm 1.49\%$ of the total flora vs. uninvaded islands, $15.2 \pm 2.13\%$; ANCOVA: Island invasion status $F_{1,13} = 21.4$, $P < 0.001$, covariate (island size) $F_{1,13} = 0.29$, $P = 0.599$; Fig. 3). These results at a whole-island, whole-flora scale contrast with results at a 10-m \times 10-m scale on the same islands, where uninvaded islands had greater numbers of non-native plant species than invaded islands (Mulder et al. 2009). This suggests that although there is a greater probability of a non-native plant species establishing on invaded islands (hence a larger total flora), those non-native plant species that do establish on uninvaded islands thrive in conditions of high nutrient concentrations and disturbance resulting from seabird populations on them (Ellis 2005).

Analyses of floras such as these can aid setting restoration goals. If the intention of restoration of an island is to have a flora similar to that of other islands in the same region, then restoration plans for islands smaller than 20 ha in northern New Zealand should note the islands are likely to be dominated by woody plants (mostly dicotyledons), with a low diversity of ferns and monocotyledon herbs (Fig 1b). Restoration plans for islands undergoing post-fire successions might suggest a role for conifers (e.g. Miller et al. 1994) because they commonly feature in post-fire successions in the warm temperate northern North Island (Ogden & Stewart 1995). However, conifers are very rare on most islands undergoing post-fire successions in this region and are usually in small populations in uncommon habitats. Examples are kauri (*Agathis australis*) which on Aorangi is confined to a single rocky knoll (de Lange & Cameron 1999) and mataī (*Prumnopitys taxifolia*) on the Hen and Chickens Islands represented by a single individual (Cameron 1984). The current rarity of native New Zealand conifers on these islands also conforms to their known ecology. High soil phosphorus (P) concentrations resulting from inputs from extant seabird colonies on islands (Fukami et al. 2006) or as legacies from extirpated colonies (Hawke et al. 1999) is expected to favour angiosperm dominance because New Zealand conifers appear to be best adapted to infertile soils (Coomes et al. 2005). Current or legacy effects of marine nutrients from seabirds may explain the rarity of conifers even on larger islands in warm temperate New Zealand. Rimu (*Dacrydium cupressinum*) is nearly absent from rain forest on Hauturu, but is a common component of forest on the adjacent mainland (Hamilton & Atkinson 1961). A possible explanation is that seabird nutrient input to soil on Hauturu (Rayner et al. 2007) results in lower competitive ability of rimu compared with species with which it co-occurs on the adjacent mainland.

An implication of the research comparing seabird-dominated islands with those affected by rats is that although eradication of seabird predators is an important step in island restoration, in cases where seabirds have been extirpated and fail to recolonise, restoration of island ecosystem processes is likely to require establishment of new seabird colonies (e.g. Miskelly & Taylor 2004). Further research is needed to determine whether ecosystem processes on islands that have undergone restoration through predator eradication recover

to the state of those of islands that have never been invaded and, if so, over what timeframe. Research is also needed to determine whether legacy effects of biological invasions are a barrier to achieving restoration of an island to the equivalent state of uninvaded islands. A further challenge is to develop an integrated approach to better understand linkages between marine ecosystems and island ecosystems. This is particularly important in the context of climate change, which is likely to increase sea surface temperature further and in turn negatively affect the marine food webs that underpin the seabird communities that breed on islands.

Changing philosophies for island management

New Zealand is now perceived internationally to be at the cutting edge of island conservation (Krajick 2005; Rauzon 2007). This reputation has been secured in part because of the application and adaptation of new technology to manage islands, notably helicopter spread of rodent bait. New Zealand island managers also developed methods to deal specifically with local problems. Two examples are the successful use of dogs for detecting individual rodents on islands and the development of new traps and lures for the detection and eradication of stoats. In 2009, New Zealanders are engaged in eradications on islands worldwide.

So far we have emphasised the technological and biological advances that have given rise to New Zealand's pre-eminence in island restoration. However over the last 20 years there has been increasing emphasis on the social and economic components that attend island management and restoration. Until the late 1980s, island management was principally the province of Government agencies and focused on the prevention of further extinctions, often through translocations of threatened taxa (Parker 2008). Twenty years later, island management and restoration has become the interest of a wide community, with various aspirations. Management of New Zealand's islands takes place increasingly in a global context. The confirmation of World Heritage status for New Zealand's subantarctic islands in 1998 is indicative of their international significance. The global distribution of the biota, such as migratory seabirds, (Shaffer et al. 2006) has implications for their conservation. For example, in 1998 an oil spill off the coast of California killed numerous seabirds, amongst which was identified a sooty shearwater banded on Codfish Island. This clear link led to compensation being paid to the Rakiura Māori community, the tangata whenua of the island. They used some of the compensation money to pay for eradication of ship rats, in 2006, from four islands on which sooty shearwaters were threatened by these predators (Harper 2007). The by-catch of seabirds in the fishing industry in New Zealand and international waters has implications for populations of seabirds on islands and is the subject of current research (Waugh 2006). Compensation from the fishing industry for marine by-catch may similarly be used for island restoration (Donlan & Wilcox 2008).

There is now increased awareness among the public of the conservation value of offshore islands (Anyan & Lang 2008). New translocations to islands or new invasions often feature as national news. A result is that management and restoration of islands is increasingly viewed independently of tenure, with comprehensive management plans prepared across archipelagos. An example is the current plan for

complementary management of all islands in the eastern Bay of Islands, with economic benefits anticipated as restored islands attract tourists. Likewise, island restoration can transcend tenure differences on individual islands. For example, goats were eradicated in 2006 from Great Barrier Island (27 761 ha; J. Craw pers. comm.), a permanently-inhabited and partially-farmed island with multiple tenures and owners. Many islands are in private ownership, and restoration of some of these has been undertaken. For example, eradication of Norway rats has taken place on the privately owned Noises Islands in the Hauraki Gulf, as part of a partnership that began in the 1980s between the owners and Government agencies.

The revegetation and translocations of land birds to Tiritiri Mātangi was achieved by a supporters group together with DOC (Rimmer 2004). This approach has been adopted as a model for programmes for the management of other islands, e.g. Motuora, Motuihe, and Mana. In 2009, the public no longer rely on central or local government to purchase and manage islands for conservation. For example, after DOC considered the environmental value of Kaikoura Island (535 ha) too low to support its purchase, public pressure resulted in a collaboration between private donors and the Government for purchase of the island in 2005 and the formation of a charitable trust to restore it and use it for environmental education. Eradication of selected non-native plants has taken place and work to eradicate all non-native mammals was underway in 2009. Restoration goals for islands vary from eradication of most non-native species on Raoul Island, to those where maintenance of non-native plant communities is desirable as habitat for threatened native taxa, such as takahē on Tiritiri Mātangi, or as a means of managing archaeological sites as on Motutapu. For Māori, maintenance of non-native plants that they introduced may also be important, for example the maintenance of non-native *Cordyline fruticosa* on two islands near Mimiwhangata (E.K.Cameron, unpublished data) is desirable for Ngātiwai, the tangata whenua, who regard this species, a relic of their former gardens on the islands, as a taonga (treasure).

The tangata whenua have a unique position in island management that stems from centuries of civilisation on many islands. For example, islands are an essential part of the way of life of Ngāti Hei, an iwi (tribe) of the eastern Coromandel Peninsula. This tribe has developed an intimate knowledge of individual islands as resources of food or shelter. Motu (islands) and their resources were a vital part of their seasonal migrations and they also offered respite from neighbouring tribes. Islands with cliffs served as impregnable coastal-watch towers from which impending threats or visits from allies could be assessed. In times of peace, islands resumed their usual role as a base for seafood gathering, gardening, bird harvesting and maintaining the mauri (life force) and ahi-kaa (continued occupation). In the 21st century, motu remain vital for Ngāti Hei and will always be vibrant reminders of the iwi's past and an integral part of their future.

Ngāti Hei place considerable importance on the restoration of island ecosystems and emphasise the need to re-establish island food webs and to support the translocation of species they regard as essential. Alongside the need to develop ecological knowledge about islands, Ngāti Hei and other Hauraki Māori place equal importance on developing this knowledge in a context of Māori culture to yield a greater understanding of their ancestors and their reasoning. In particular, past traditional management ensured sustainability of a certain species for sustenance rather than preservation. For this reason, current restoration of islands requires an understanding of the mechanisms of rāhui (no harvest) and tapu that were used in

the past to restore animals to safe numbers. Mātauranga Māori provides valuable indicators of past populations and trends, knowledge that might guide restoration and be used to support scientific studies, in what might be described as a “responsible collaboration”. Restoration of islands is also seen as a step towards regaining mauri which has diminished through the lack of use of traditional food gathering methods. Therefore, a key intent in the restoration of Ohinau (43 ha) by Ngāti Hei is that it becomes a wānanga (a place of learning), in which two knowledge sets – Mātauranga Māori and science – inform visitors and whanau (extended families) who seek to learn more about the motu.

Into the future

Where might we be in another 20 years in terms of island restoration? The frequency of eradications of non-native mammals from islands is likely to decline because they have already been achieved from many priority sites. However, the total area cleared of non-native mammals will probably continue to increase as eradications in larger and more complex systems are attempted. Ever-increasing numbers of non-native plants and invertebrates will continue to be a challenge to control. The next two decades could productively become a period of consolidation. With more emphasis on recording the responses of biotic communities and ecosystems, we should have a much clearer view of what has been achieved through past eradications and current restoration initiatives. We should also have far more robust predictions of what is achievable; perhaps to the extent of answering some of the challenges associated with “reconstructing the ambiguous” (Simberloff 1990a). The rate of progress has already captured the public's imagination. However, if the reasons for this interest are to be quantified, far more needs to be known about the social and economic effects of various forms of island management. With better understanding of the combined biodiversity, social and economic benefits that are possible, we may then be equipped to take on the restoration of larger and more challenging islands such as Great Barrier Island and Rakiura (174 600 ha). A vision for the restoration of Great Barrier Island has already been proposed (Ogden & Gilbert 2009). Technical challenges for its restoration can probably be overcome; obtaining a unified view among a diverse human population on this permanently-inhabited island is likely to be the main determinant of the success of a project on this scale.

New Zealand's achievements in island management are likely to be held in high regard internationally in future. Perhaps the most appropriate recognition of these achievements would be to complete the formal nomination of islands currently on the Government's World Heritage short list. The north-eastern islands and the Kermadec Islands would then join Fiordland and subantarctic islands which already have World Heritage status. Formal nomination would emphasise that the Government, iwi and New Zealand public still see island management as something that is being achieved extraordinarily well and which meets a wide variety of social and biodiversity goals. There may be no better accolade than to demonstrate island management that is sufficiently enlightened to meet the core criterion for World Heritage sites: outstanding universal value.

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Appendix 1. List of islands mentioned in the text with names according to New Zealand Geographic Board, latitude and longitude, and (for individual islands) areas from Atkinson IAE, Taylor RH 1992 Distribution of alien animals on New Zealand islands. Investigation No 547. DSIR Land Resources Contract Report No. 92/59

Name	Location	Area ha	Name	Location	Area ha
Aorangi, Poor Knights Islands	35.48 S, 174.74 E	110	Mayor Tuhua	37.28 S, 176.24 E	1277
Auckland	50.40 S, 166.05 E	45975	Mokohinau Islands	35.93 S, 175.09 E	
Atiu or Middle, Mercury Islands	36.64 S, 175.86 E	13	Mokoia	38.08 S, 176.29 E	135
Bounty Islands	47.42 S, 179.03 E		Motuhoropapa, Noises Islands	36.69 S, 174.96 E	8
Breaksea	45.58 S, 166.64 E	170	Motuihe	36.81 S, 174.94 E	195
Brothers Islands northern	41.11 S, 174.44 E	4	Motuora	36.50 S, 174.80 E	86
Campbell	52.53 S, 169.18 E	11216	Motutapu	35.19 S, 174.00 E	1560
Chalky	46.05 S, 166.52 E	475	Moutohora	37.85 S, 176.97 E	173
Codfish Whenuahou	46.78 S, 167.62 E	1396	Ohinau	36.73 S, 175.88 E	43
Cuvier Repanga	36.43 S, 175.77 E	170	Otata, Noises Islands	36.69 S, 174.97 E	15
Eastern Bay of Islands	35.21 S, 174.23 E		Poutama	47.27 S, 167.41 E	36
Great Barrier Aotea	36.10 S, 175.41 E	27761	Rangitoto	36.80 S, 174.86 E	2321
Great Manawa Tawhi	34.16 S, 172.14 E	408	Rangitoto ki te Tonga D'Urville	40.82 S, 173.85 E	16782
Hauturu/Little Barrier	37.21 S, 175.89 E	3083	Raoul, Kermadec Islands	29.27 S, 177.93 W	2938
Hen and Chickens Islands	35.92 S, 174.73 E		Resolution	45.65 S, 166.75 E	20860
Jacquemart, Campbell group	52.62 S, 169.12 E	19	Snares Islands	48.02 S, 166.35 E	
Kaikoura Selwyn	36.18 S, 175.31 E	535	South East Rangatira,		
Kapiti	40.85 S, 174.92 E	1970	Chatham Islands	44.35 S, 176.17 W	219
Korapuki, Mercury Islands	36.66 S, 175.85 E	18	Stephens Takapourewa	40.67 S, 173.99 E	150
Limestone	35.78 S, 174.36 E	38	Stewart/Rakiura	47.05 S, 168.05 E	174600
Long	41.12 S, 174.28 E	142	Taukihepa/Big South Cape	47.24 S, 167.40 E	939
Mahuki Anvil	36.23 S, 175.29 E	45	The Aldermen Islands	36.97 S, 176.08 E	
Mana	41.09 S, 174.79 E	217	Tiritiri Matangi	36.61 S, 174.89 E	196
Mangere, Chatham Islands	44.27 S, 176.30 W	113	Ulva	46.93 S, 168.13 E	259
Maud	41.02 S, 174.28 E	309	Whakaari/White	37.52 S, 177.18 E	313

Appendix 2. Biodiversity responses and conservation on islands after pest eradication by ground-based operations before 1985, with responses identified as natural recovery (recovery), translocation, intensive management of small populations (management) and management of resident populations within the site (enhancement)

Location	Invasive species removed	Response and achievements
Manawa Tawhi (Great Island, Three Kings Group)	Goats	> 200 species of plants; up to 30 species of endemic snails including <i>Placostylus bollonsi</i> (recovery)
Aorangi (Poor Knights Group)	Pigs	18 species of plants, 5 species of snails, 13 species of insects, 6 species of reptiles, 2 species of birds (recovery)
Takapourewa (Stephens Island, Cook Strait)	Cats	Cook Strait tuatara, fairy prions (recovery), Stephens Island frog (enhancement)
Maud Island (Marlborough Sounds)	Pigs, goats	Kakapo, takahe (management); Cook Strait giant wētā, speckled skink, orange-fronted parakeet, fluttering shearwater (translocation); Maud Island frog (enhancement)
Titi Island (Marlborough Sounds)	Norway rats	Brothers Island tuatara, Cook Strait giant wētā, flax weevil (translocation)
Hawea (Fiordland) Norway	Rats	Fiordland skink (recolonisation + translocation)
Rangatira (South East Island, Chatham Islands)	Goats	Black robin (translocation)
Mangere Island (Chatham Islands)	Cats, rabbits	White-faced storm petrel, black-winged petrel (recolonisation); black robin, Chatham Island snipe, Chatham Island tomtit, shore plover (translocation)

Appendix 3. Biodiversity responses and conservation on islands after pest eradication operations after 1985, with eradication method identified as ground-based (g) or by aerial spread (a). The list excludes species translocations or other management where pest eradication was not required.

Location	Invasive species removed and method	Response and achievements
Raoul Island (Kermadec group)	Goats (g), kiore and Norway rats (a), cats (a, g)	Forest recovery: post goats. Endemic shrub <i>Hebe breviracimosa</i> (regeneration); black-winged petrel, Kermadec petrel, wedge-tailed shearwater, Kermadec parakeet, red-tailed tropic bird, sooty tern, spotless crake (recolonisation); land crab, ghost crab (reappearance)
Motuopao (Cape Maria van Dieman)	Kiore (g)	Matapia gecko, robust skink (translocation); flax snail (recovery)
Coppermine (Marotere group, Hen and Chickens)	Kiore (g-failed) (a)	Pycroft's petrel, little shearwater, tuatara, Suter's skink, Duvaucel's gecko (recovery), Mokohinau skink (translocation)
Lady Alice (Marotere group, Hen and Chickens)	Kiore (a)	Pycroft's petrel, little shearwater, tuatara, Duvaucel's gecko (recovery); McGregor's skink, Mokohinau skink (x2), large darkling beetle, karo weevil (), carnivorous snail (translocations); Pacific gecko (reappearance+translocation); North Island kokako (management)
Whatupuke (Marotere group, Hen and Chickens)	Kiore (a)	Tuatara, ornate skink, moko skink, Suter's skink, Duvaucel's gecko (recovery); McGregor's skink, Mokohinau skink (translocation); Pacific gecko, copper skink (reappearance)
Hauturu (Hauraki Gulf)	Cats (g), kiore (a)	Black petrel (enhancement); North Island saddleback, kokako (translocation); kakapo (management): post cat. Cook's petrel (recovery); tuatara (enhancement); chevron skink, Duvaucel's gecko (reappearance): post kiore
Tiritiri Mātangi (Hauraki Gulf)	Cat (g), kiore (a)	Red-crowned parakeet, saddleback, brown teal, North Island fernbird, whitehead, North Island robin, little spotted kiwi (translocation); takahe (management): post cat. common gecko (reappearance); tuatara, Duvaucel's gecko, shore skink (translocation); stitchbird, kokako (management): post kiore
Pakatoa (Hauraki Gulf)	Norway rat (g)	North Island weka (translocation)
Motuihe (Hauraki Gulf)	Norway rat, mice (a), cat (g), rabbit (a+g)	New Zealand dotterel (recovery); North Island saddleback, red-crowned parakeet (translocation)
Cuvier (E. Coromandel)	Goat, cat (g), kiore (a)	North Island saddleback, red-crowned parakeet (translocation): post goat and cat. tuatara (recovery and enhancement), milktree (<i>Streblus banksii</i>) (regeneration), fluttering shearwater (recolonisation), Mercury Island tusked wētā, Pycroft's petrel (translocation): post kiore
Double (Mercury Islands)	Kiore (g)	Plants (recovery); Mercury Island tusked wētā (translocation)
Korapuki (Mercury Islands)	Kiore, rabbit (g)	Large darkling beetle, tree wētā, Mercury Island tusked wētā, robust skink, marbled skink, Whitaker's skink, Suter's skink (translocation); honeydew scale, common gecko, Duvaucel's gecko, copper skink, shore skink, diving petrel, kingfisher (recovery)
Red Mercury (Mercury Islands)	Kiore (a)	Tuatara (enhancement); Mercury Island tusked wētā, robust skink, Whitaker's skink (translocation)
Stanley (Mercury Islands)	Kiore, rabbit (a)	Tuatara (enhancement); Mercury Island tusked wētā, robust skink, Whitaker's skink (translocation)
Ohinau (Ohinau Islands)	Mice, kiore, rabbit (a)	Mercury Island tusked wētā (translocation)
Mahurangi (Hahei)	Norway rats (g)	Mohoenui wētā (translocation)
Tuhua (Mayor)	Pigs (g), Norway rats, kiore, cats (a)	North Island robin, brown teal, North Island brown kiwi
Moutohora (Whale)	Goats (g), Norway rats, rabbits (a+g)	Tuatara, North Island saddleback, North Island brown kiwi (translocations); grey-faced petrel (recovery)
Mokoia (Lake Rotorua)	Goats (g), Norway rats (g), mice (a)	North Island robin, saddleback, North Island brown kiwi
Kapiti (West Wellington coast)	Axis deer, goats, cats, pigs, brush-tailed possums (g), Norway rats, kiore (a)	North Island saddleback, North Island kokako (translocation); takahe, stitchbird (management)

Mana (South Wellington coast)	Mice (a+g)	Cook Strait giant wētā, common gecko, McGregor's skink (recovery); brown skink (reappearance); flax weevil, speargrass weevil, Wellington green gecko, speckled skink, spotted skink, diving petrel, fluttering shearwater, fairy prion, yellow-crowned parakeet, North Island robin, brown teal, shore plover (translocations); takahe (management)
Matiu/Somes (Wellington Harbour)	Ship rat (g)	Common gecko, spotted skink (recovery); Cook Strait giant wētā, Wellington tree wētā, Brothers tuatara, forest gecko, ornate skink, red-crowned parakeet, North Island robin (translocation)
Nukuwaiata (Marlborough Sounds)	Pigs (g), kiore (a), weka (a+g)	Nikau, titoki, kohekohe (regeneration); spotted skinks (recovery); Hamilton's frog, mohua (translocation)
Te Kakaho (Marlborough Sounds)	Weka (g)	Brown skink (recovery)
Allports (Marlborough Sounds)	Mice, possums (g)	South Island robin
Long (Marlborough Sounds)	Pacific rat (a)	Little-spotted kiwi, South Island saddleback, parakeet (sp?), <i>Leiopelma pakeka</i>
Motuarua (Marlborough Sounds)	Pacific rat (g)	Okarito brown kiwi, Marlborough green gecko?, South Island robin, South Island saddleback, <i>Leiopelma pakeka</i>
Whakatere-Papanui (Marlborough Sounds)	Norway rat, Pacific rat (a)	Tuatara, Marlborough green gecko, Cook Strait giant wētā
Breaksea (Fiordland)	Norway rat (g)	Flax weevil, knobbed weevil, South Island saddleback, mohua (translocations); Fiordland skink (recolonisation)
Anchor (Fiordland)	Stoat (g)	Rock wren (management)
Chalky (Fiordland)	Stoat (g)	Kakapo (management), orange-fronted parakeet (translocation)
Codfish (Whenuahou) (Stewart Island)	Weka, brush-tail possum (g), Pacific rat (a)	Cook's petrel, fairy prion (recovery); kakapo, Campbell Island teal (management)
Ulva (Stewart Island)	Norway rat (g)	Common skink, Stewart Island fernbird, mohoua (translocation)
Putauhina (Stewart Island)	Pacific rat (a)	Diving petrel (recolonisation)
Campbell	Cattle, sheep (g), Norway rat (a)	Pipit (recovery); snipe, white-chinned petrel, grey-backed storm petrel (recolonisation); Campbell Island teal (translocation)

Appendix 4. Data sources for Figs 2 and 3:

Invaded islands: Motukaramarama, Motuoruhi, and Motutapere (Esler 1978 *New Zealand Journal of Botany* 16: 25–44 and Wilcox 2005 *Auckland Botanical Society Journal* 60: 1–12), Motuhoropapa and Otata (both Cameron 1998 *Auckland Botanical Society Journal* 53: 25–35), Goat Island (Motu Hawere) (Gordon & Ballantine 1977 *Tane* 22 Supplement; Benham 1996 *Auckland Botanical Society Journal* 51: 49–52 and E.K. Cameron, unpublished data); Aiguilles and Motueka (both E.K. Cameron, unpublished data);

Uninvaded islands: Aorangi (de Lange and Cameron 1999 *New Zealand Journal of Botany* 37: 433–468), Green Island (Atkinson 1964 *New Zealand Journal of Botany* 2: 385–402), Middle Island (Atkinson 1964 *New Zealand Journal of Botany* 2: 385–402, Cameron 1990 *Journal of the Royal Society of New Zealand* 20: 273–285), Ruamāhuaiti and Ruamāhuanui (Cochrane 1962 *Transactions of the Royal Society of New Zealand – Botany* 1: 331–341, Court et al. 1981 *Tane* 27, 41–60 and P.J. Bellingham and E.K. Cameron, unpublished data), Aorangaia, Archway, and Ohinauiti (all E.K. Cameron, unpublished data).