Methods for monitoring invertebrate response to vertebrate eradication

M. Houghton[1], A. Terauds[2] and J. Shaw[1]

[1]Centre for Biodiversity and Conservation Science, School of Biological Sciences, The University of Queensland, Brisbane, QLD 4072, Australia. <m.houghton@uq.net.au>. [2]Antarctic Conservation and Management, Australian Antarctic Division, Department of the Environment, 203 Channel Highway, Kingston, TAS 7050, Australia.

Abstract Once an island vertebrate eradication is deemed successful, it is typically assumed that ecosystem recovery will follow. To date, most post-eradication monitoring focuses on the recovery of key threatened or charismatic species, such as seabirds. Little attention has been given to monitoring and quantifying the response of invertebrate communities. Rabbits (*Oryctolagus cuniculus*), house mice (*Mus musculus*), and ship rats (*Rattus rattus*) impacted sub-Antarctic Macquarie Island for over 140 years, with wide ranging ecosystem impacts. In 2014, the eradication of rabbits and rodents was officially declared successful. To determine whether management objectives are being met, we are investigating the response of invertebrate communities to pest eradication, using both historic data and contemporary surveys to track changes over space and time. To achieve this, we have developed a survey strategy that is effective and efficient. Here we report on the merits of utilising a variety of invertebrate trapping methodologies to establish current baselines for future invertebrate monitoring. We identify sampling techniques that are most effective for specific groups of taxa, particularly those of interest to post-eradication monitoring, and how the implementation of such methods can improve and facilitate effective post-eradication monitoring of invertebrates.

Keywords: baselines, conservation, insects, island, rabbits, restoration, rodents, sub-Antarctic

INTRODUCTION

Island invertebrates are impacted by invasive species, particularly on remote, unpopulated islands in the Southern Ocean (Chown, et al., 2008; Angel, et al., 2009; St Clair, 2011: Russell, 2012; Thoresen, et al., 2017). Non-native plants and invertebrates have been unintentionally introduced to Southern Ocean Islands (SOI) (Frenot, et al., 2005; Chown, et al., 2008; Convey & Lebouvier, 2009). Non-native vertebrates have also been introduced, both intentionally and inadvertently. For example, rabbits (*Oryctolagus cuniculus*), cats (*Felis catus*), dogs (*Canis lupus familiaris*), sheep (*Ovis aries*), goats (*Capra aegagrus hircus)*, weka (*Gallirallus australis*), pigs (*Sus scrofa domesticus*), brown trout (*Salmo trutta*) and reindeer (*Rangifer tarandus*) were all intentionally introduced to SOI, whereas ship rats (*Rattus rattus*), brown rats *(Rattus norvegicus)* and house mice (*Mus musculus*), were unintentional introductions (Copson & Whinam, 2001; Courchamp, et al., 2003; Convey and Lebouvier 2009; McGeoch, et al., 2015). Grazing by non-native vertebrates on SOI has led to invertebrate habitat modification (Vogel, et al., 1984; Chapuis, et al., 2004), and direct predation by rodents has severely modified and depleted invertebrate populations (Copson, 1986; Chown & Smith, 1993; Angel, et al., 2009; St Clair, et al., 2011; Russell, 2012; Treasure, et al., 2014).

Macquarie Island (54.6208° S, 158.8556° E) lies 1,500 km south-east of Tasmania, Australia. The island is a World Heritage area managed as a Nature Reserve by the Tasmanian Parks and Wildlife Service (Copson & Whinam, 2001). Discovered in 1810, the island's early human history involved seal harvesting (elephant seals, *Mirounga leonina;* fur seals *Arctocephalus pusillus, A. forsteri, A. tropicalis*) and penguin harvesting (king penguins, *Aptenodytes patagonicus* and royal penguins, *Eudyptes schlegeli*). Many non-native species of flora and fauna were introduced during this time, both intentionally and inadvertently. Ongoing control of cats and rabbits by various methods and at varying levels of effort led to fluctuating populations (Robinson & Copson, 2014; Terauds, et al. 2014). Consequently, native fauna and vegetation were impacted by varying levels of predation and grazing (Scott & Kirkpatrick, 2008; Scott & Kirkpatrick, 2013; Bergstrom, et al., 2009; Whinam, et al., 2014). Over time, island managers have removed almost all invasive vertebrate species from Macquarie Island (Copson & Whinam, 2001), the most recent being cats in 2000 (Robinson & Copson, 2014) and rabbits and rodents in 2014 (Springer, 2016). The latter were the target of the Macquarie Island Pest Eradication programme, which was the largest multi-species project of its kind at the time, costing AU$24.5 million. The project's overall objective was to '…restore Macquarie Island biodiversity and geodiversity to a natural balance – free of the impacts of introduced pest species… [with] …vegetation, seabird and invertebrate populations recovered to levels naturally supported by the environment' (Parks and Wildlife, 2009). We developed a study to assess the success of this project for invertebrates; specifically, to better understand if they have 'recovered' following removal of mammalian herbivores and predators, using both historic data and contemporary surveys.

Invertebrates play a key role in ecosystem function (Kremen, et al., 1993; Hutcheson, et al., 1999; Gerlach, et al. 2013). They drive nutrient-cycling and decomposition processes on SOI (Smith & Steenkamp, 1990; 1992; Smith, 2007; 2008). Thus, establishing a baseline and measuring their response to ecosystem change informs the state of the island ecosystem. Many types of invertebrates are useful proxies for assessing ecosystem change, reflected in their species richness, species turnover and community composition (Kremen, et al., 1993; Hutcheson, et al., 1999; Towns, et al., 2009). Indicator taxa are particularly useful in monitoring the effects of habitat management at the ground layer (e.g. ants, millipedes, snails, ground beetles, some spiders), in foliage (e.g. ants, leaf beetles, some spiders and moths), and in open habitats (e.g. ants, crickets, grasshoppers, and butterflies) (Gerlach, et al., 2013). Moreover, their high density, short life span, ubiquity and rapid response to changing environmental conditions, make invertebrates ideal for long-term monitoring (Samways, et al., 2010; McGeoch, et al., 2011).

Despite their suitability as indicators monitoring of invertebrates post-eradication is rarely undertaken and their response to eradication is infrequently determined.

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 381–388. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Developing appropriate survey methods and sampling strategies is crucial for a monitoring programme. Here we test a variety of invertebrate survey techniques and report on the merits of using specific invertebrate trapping methodologies to establish baselines for future invertebrate monitoring and to facilitate comparisons with previous surveys. Our recent surveys included most of the invertebrate trapping techniques previously employed on the island by historical surveys. Our survey design aimed to measure invertebrate response to vertebrate eradication and vegetation rehabilitation, track change in invertebrate community composition and numbers, and establish baselines for future monitoring. Specifically, our objectives in this paper, are to 1) compare the efficacy of using different invertebrate trap types in achieving monitoring objectives, 2) assess the effectiveness of historical trapping methods in informing contemporary survey design, and 3) investigate the benefits and limitations of using historical data for tracking changes over time. We also discuss how choosing appropriate methods is a key process for effective and efficient post-eradication monitoring of invertebrates.

METHODS

Survey design

Determining invertebrate community changes over time requires definitive and repeatable methods and detailed site information. Our experimental design (i.e. our choice of survey/trapping techniques and site selection) was informed by analysing invertebrate trapping experimental designs, methods and results from historical surveys on Macquarie Island. Following a thorough review of the scientific literature five key resources were selected to inform our experimental trapping design and methods: Watson (1967), Greenslade (1987), Anonymous (1993–94, reported in Stevens, et al., 2010), Davies & Melbourne (1999), and Stevens, et al., (2010). Each of these historical surveys utilised different combinations of methods (Table 1). Based on this information, the following survey methods were tested in our study: pitfall traps, sweep netting, litter extraction, and timed hand collecting (referred to as '20-minute counts').

Site selection

For this study, sampling was carried out at ten historic and ten new sites (Fig. 1). This provided 20 sites in total for the 2015/16 post-eradication survey. The ten new sites were selected to ensure broader island coverage and survey additional vegetation communities across the five dominant vegetation structures on Macquarie Island (based on Selkirk, et al., 1990) – feldmark (plateau), lower coastal slopes dominated by *Stilbocarpa polaris* (Macquarie Island cabbage), tall grassland (tussock) dominated by *Poa foliosa*, short grasslands (including *Deschampsia* spp., *Festuca contracta, Agrostis magellanica, Luzula crinita, Uncinia* spp.), and herbfield dominated by *Pleurophyllum hookerii*. Most sites were heavily impacted by rabbits in the past (Bergstrom, et al., 2009; Whinam, et al., 2014). In total, four *Stilbocarpa polaris* sites were surveyed in 2015/16, three short grassland sites, five tall grass sites, four herbfield sites, and four feldmark sites.

Sampling techniques

Five pitfall traps were established at each of the 20 sites, in a line transect along a recorded bearing, five metres apart. Expert advice from the Tasmanian Department of Primary Industries, Parks, Water and the Environment (M. Driessen, pers. comm.), informed the pitfall trap preparation, spacing, pattern of positioning, and preservative used. Pitfall traps were constructed of straight sided, plastic jars 7 cm in diameter, approximately 7 cm deep, with ca. 1 cm of 100% propylene glycol preservative added. Pitfall diameter was selected based on other studies

Table 1 Trapping methodology employed during invertebrate sampling studies on Macquarie Island – Watson in 1961 (reported in Watson, 1967), Greenslade in 1986–87 (reported in Greenslade, 1987), Anonymous in 1993–94 (reported in Stevens, et al., 2010), Davies and Melbourne in 1996 (reported in Davies & Melbourne, 1999), Stevens, et al., in 2009–10 (reported in Stevens, et al., 2010).

	Watson 1961	Greenslade 1986–87	Anonymous 1993–94	Davies & Melbourne 1996	Stevens, et al., 2009–10
Length of sampling	Year-long	December – January	Year-long	February –March	October –January
Extent of sampling	Island-wide	Northern sites	Northern sites	Island-wide	Northern sites
# Sites	Not specified	8	4	67	41
# Pitfalls/site	0	5	Not specified	3	3
# Pitfall trap days	-	5–20	30	ca. 42	7–21
Pitfall diameter (cm)	-	1.8	'Large' & 'small'	3	'Large' & 'small'
Pitfall medium	-	Ethanol	Not specified	Ethylene glycol/ detergent	Ethanol
# Yellow pan trap/site	Yes, # not specified	0	0	1	0
Vegetation Beating	No	Yes, # not specified	No	No	Yes, # not specified
Vegetation Sweeping	Yes, # not specified	Yes, # not specified	No	No	No
Litter volume (Lt)	Yes, # not specified	2–4	0	0	1 over 1 m²
Litter extraction method	Berlese funnels	Berlese funnels	n/a	n/a	Berlese funnels
# Soil Cores	Yes, # not specified	11–16	5	0	0
20-minute counts (hand collecting)	Yes (not timed)	Yes (not timed)	No	Yes	No

that have proven the effectiveness of larger trap sizes (Brennan, et al., 1999; Ward, et al., 2001; Work, et al., 2002; Woodcock, 2005). Propylene glycol was chosen a preservative due to it being environmentally benign, highly viscous and slow to evaporate. Pitfall trap holes were dug with a soil corer to ensure a snug fit and the pitfall rim was flush with the ground surface. Where necessary, a small amount of vegetation was cleared from the trap rim. At the 20 sites, pitfall traps were collected approximately every 10 days between October and December and reset upon collection for a total period of up to ca. 42 days. Further pitfall sampling was repeated monthly in January, February and March for approximately 5–10 days at eight sites.

For litter sampling, at each site, a 1m² quadrat was used to define the collection area, and three collections were made of one litre of litter at each site. Litter was transported back to the station laboratory for sorting and invertebrate extraction within a maximum of three days from collection. In feldmark sites where litter was scarce, litter collection was over 4 m² and up to 1 litre of material – the exact area and volume was recorded.

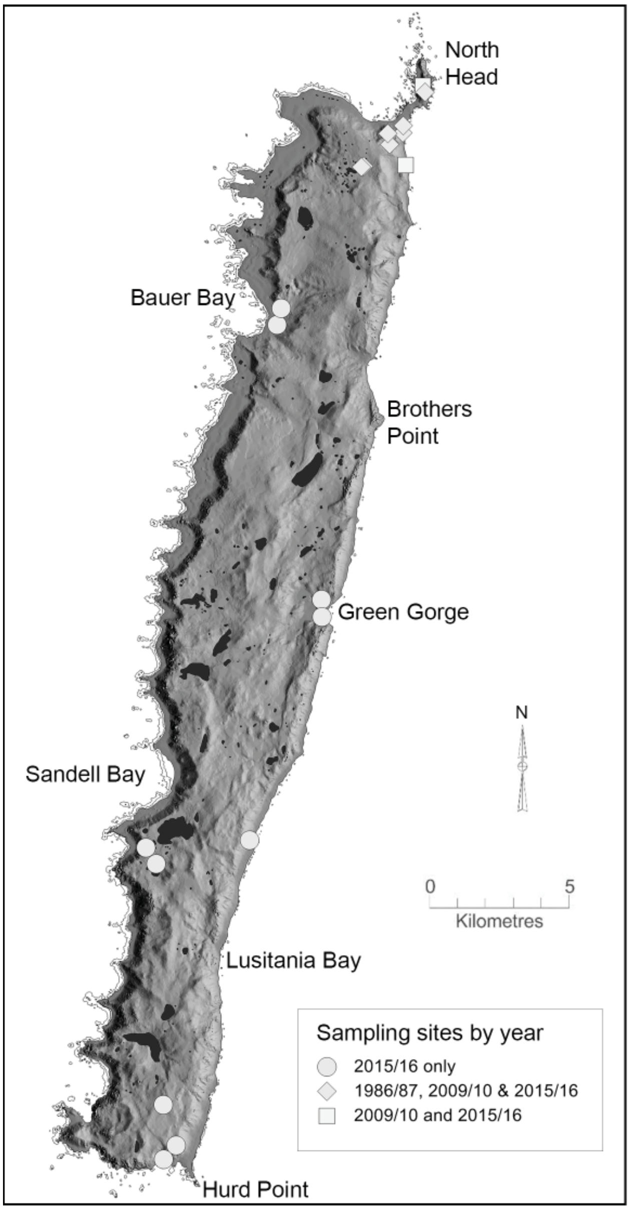

Fig. 1 Map of 20 invertebrate trapping sites surveyed at Macquarie Island in 2015/16. All historic sites sampled in 1986/87 (indicated by grey diamonds) were resampled in 2015/16.

Timed counts of 20 minutes were conducted at least twice over the study period at each site, involving focussed searching with an aspirator tube and tweezers, collecting all invertebrates encountered, particularly at the base of vegetation and under stones.

Sweeping of vegetation tops with nets required dry conditions with light winds. Hence, sweeping was conducted opportunistically, at a minimum of twice at each site over the study period, in all vegetation types regardless of the canopy (i.e. also in feldmark), by walking slowly and dragging the net across the vegetation tops 30 times, on three different trajectories in the site area per sampling event.

One temperature logger i-button was installed at each of the 20 sites to monitor microclimate, at the first pitfall trap of each transect. They were attached approximately 10 cm above the ground surface on a stake with a protective housing. At each site, site-specific features such as aspect, altitude, landscape features, vertebrate fauna presence and vegetation were noted.

Processing and identification

All samples were transferred promptly to 100% ethanol and transported back to the Australian Antarctic Division for identification and storage. Using a dissecting microscope, specimens were counted and identified to species where possible, except for Acarina, Annelida, and Nematoda, which were identified to Class or Phylum level.

Data analysis

We undertook preliminary comparisons of the 2015/16 survey data on Order richness and diversity with data from historical sites established in 1986/87 (Greenslade, 1987). We calculated taxonomic richness and diversity of invertebrates in different trap types, vegetation groups and in historical data. For these purposes we pooled data from different sites to obtain the total number of taxa trapped in different trap types and vegetation groups. Invertebrate richness was calculated by summing the total number of invertebrate Orders recorded in the trap type or vegetation group of interest. Simpson's Index of Diversity (SID) was selected to compare diversity, as it takes into account both abundance and richness in each habitat. We compared the Order richness and diversity of our pitfall traps to seven historical sites and also quantified changes in abundance for three target groups (beetles, spiders and moths), using a subset of the historic and contemporary pitfall data.

We analysed data at the level of Order/Class/Phylum (hereafter referred to as 'Order') to facilitate preliminary comparison with historic data sorted to Order resolution. For analysis, larval stages and adults were grouped together for Lepidoptera, Thysanoptera, Coleoptera and Diptera.

RESULTS

Contemporary survey

Our preliminary results from the 2015/16 survey demonstrated that pitfall traps collect the largest number of individuals – in particular, Collembola (Table 2). Even when Collembola were removed from the analysis, pitfalls still collected more individuals than other trapping methods. Despite the abundance of invertebrates in pitfalls, there was considerable variance in the nature and abundance of taxa caught by the different trapping methods, with some methods proving more effective for specific taxa than others (Table 2).

Richness (number of different orders caught) and diversity (SID – richness combined with the relative

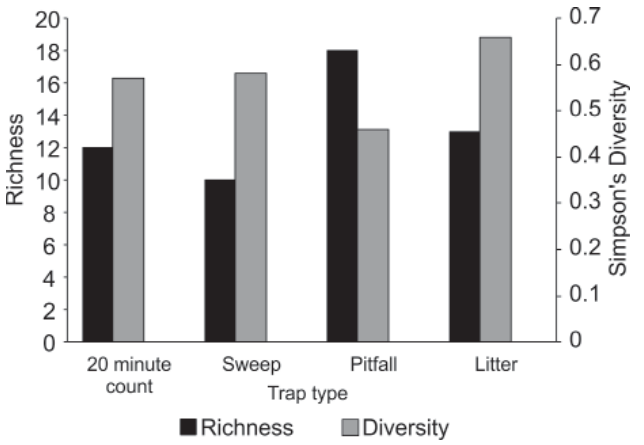

Fig. 2 Order richness (summed across 20 sites) and Simpson's diversity of four different trapping methods on Macquarie Island in 2015/16 following mammal eradication.

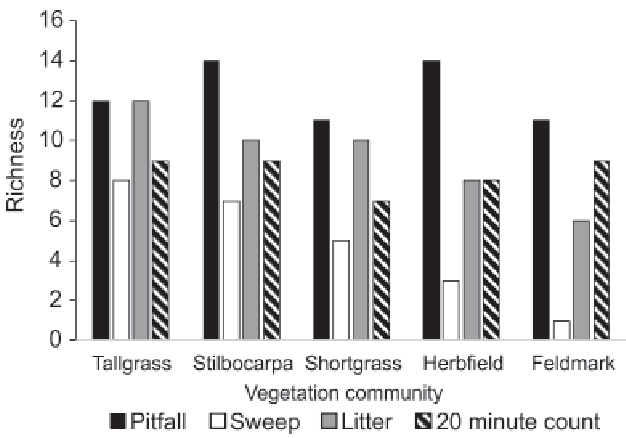

Fig. 3 Order richness (summed across 20 sites) of four trapping types in five vegetation communities on Macquarie Island in 2015/16 following mammal eradication.

abundance of the different orders caught) varied between trapping methods (Fig. 2). The SID demonstrated that although pitfalls traps yielded the greatest richness, they also had the lowest diversity, attributable largely to the dominance of Collembola. Conversely, sweeping had relatively low species richness but high SID, an indication of the greater relative abundance of different taxa trapped.

Pitfalls collected the most species regardless of habitat type (Table 2, Fig. 3). Effectiveness of the other trap types varied by vegetation community (Fig. 3). Sweeping vegetation was far more effective in tall grassland and *S. polaris*, which are often characterised by dense protective foliage, than for herbfield and feldmark vegetation, which typically have more prostrate, sparsely distributed plants. Litter collection also yielded high relative Order richness, particularly in tall grassland and short grassland vegetation communities. Twenty-minute counts were effective in feldmark, where richness was proportional to effort. The low number of taxa in this habitat were found more readily through this method of focused searching (disturbing stones and turf), than via passive pitfall trapping or surface litter collection.

The SID of pitfall trap samples across vegetation types was almost the inverse of their richness (Fig. 4). Across all vegetation types (except for feldmark), pitfall trapping diversity was much lower than for other trap types; a likely reflection of the dominance of the Collembola in pitfall traps except in feldmark. For short grassland, litter sampling proved to be exceptionally diverse. Interestingly, although taxonomic richness of sweeping in herbfield was relatively low, SID was high. Across all vegetation types, 20-minute counts were almost equal in diversity.

Table 2 The number of individuals from each Order of invertebrates collected via four different trapping methods on Macquarie Island following mammal eradication: pitfall traps, sweeping, 20-minute counts, and litter collection in the 2015/16 season following mammal eradication.

Order	Common Name	Pitfalls	Sweep	20 minute count	Litter
Gastropoda	Snails/slugs	935	2	1,294	1,019
Psocoptera	Booklouse	44	2	0	129
Hemiptera	Aphids/Bugs	3	0	0	0
Thysanoptera	Thrips	144	21	4	4
Coleoptera	Beetles	2,512	2	12	240
Diptera	Flies	945	61	51	61
Lepidoptera	Moths	3	0	4	8
Hymenoptera	Wasps	1	0	0	0
Isopoda	Crustacea	209	1	207	636
Araneae	Spiders	2,467	40	169	380
Platyhelminthes	Flatworms	20	0	1	1
Annelida	Worms	284	3	493	1,489
Copepoda	Copepods	3,615	0	2	8
Tardigrada	Tardigrades	69	0	0	0
Acarina	Mites	5,219	40	108	340
Siphonaptera	Fleas	1	0	0	1
Nematoda	Nematodes	19	0	0	0
Collembola	Springtails	43,641	277	3,609	5,040
	TOTAL	60,131	449	5,954	9,356

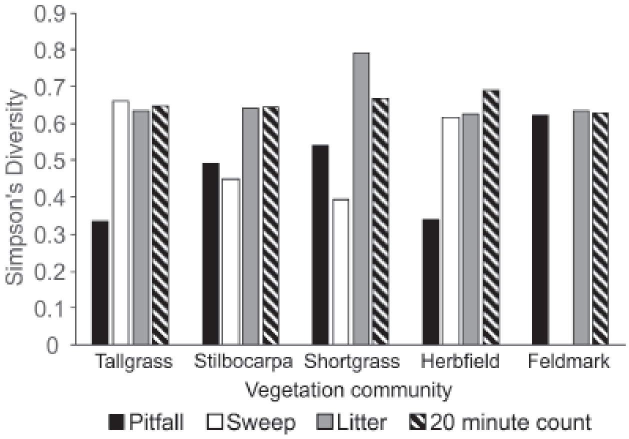

Fig. 4 Simpson's Index of Diversity (Order) of four trap types in five vegetation communities on Macquarie Island in 2015/16 following mammal eradication.

Comparisons with historical surveys

Preliminary comparisons of our data on Order richness and diversity data from the 1986/87 sites (Fig. 5) indicate considerable changes in invertebrate communities since the earlier surveys. Both Order richness and SID were generally lower during the earlier sampling period compared to 2015/16, with the exception of the feldmark F2 site, where 1986/87 samples were more speciose and more diverse. Diversity in the tall grassland site P2 and herbfield H1 were also lower in 2015/16 sampling, though richness was much higher.

Mouse prey species that were predicted to respond favourably to mouse removal, such as Coleoptera (beetles), Lepidoptera (moths) and spiders (Araneae), were trapped via pitfalls in 1986/87 and again in 2015/16 at seven sites across five vegetation types (Table 3). Coleoptera abundance was inconsistently higher in 1986/87, whereas Araneae were trapped in much higher numbers during the 2015/16 sampling. Lepidoptera were rarely trapped in both sampling events, present only in the feldmark F2 site.

DISCUSSION

When monitoring ecosystem responses following an eradication, it is critical to first identify the objectives of the management intervention. In this case, the facilitation of the "recovery" of macro-invertebrates on Macquarie Island was explicit. However, no mechanisms were put in place to assess the success (or otherwise) of this objective. Here, our preliminary study tackles the issue of how to effectively survey a suite of invertebrate species on a Southern Ocean island to detect recovery and response of invertebrates after an eradication event, and informs the selection of appropriate survey methods for specific species.

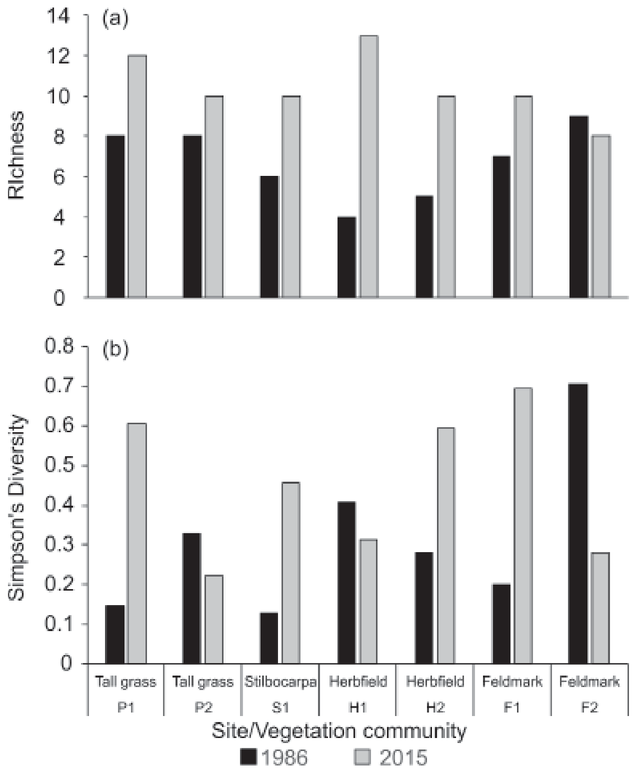

Fig. 5 (a) Order richness (summed across 20 sites) and, (b) Simpson's Index of Diversity of pitfall trapping (Order level) at seven invertebrate monitoring sites at Macquarie Island that were first sampled in 1986/87 (prior to mammal eradication) and repeat sampled in 2015/16 (post mammal eradication).

One of the clearest findings of our study was that pitfall traps collect the greatest abundance and richness of invertebrates, particularly Collembola, although they were the least diverse. Despite the difficulties in comparing abundance and sampling effort across different techniques (for example, the longer deployment time of pitfall traps compared to other trapping techniques), it is apparent that different trapping methods are more effective at capturing certain taxonomic groups. This is based on the functional traits, behaviours and preferred habitats of different taxa. For example, Psocoptera were primarily collected from litter samples, as they are detritivores with a preference for damp conditions under vegetation (Greenslade, 2006). However, some were also collected during vegetation sweeping, where they occur in smaller numbers under leaves (Greenslade, 2006). Tardigrades and Copepods

Table 3 Abundance of Coleoptera, Lepidoptera and Araneae in pitfall traps sampled at seven sites at Macquarie Island in 1986/87 (prior to mammal eradication) and 2015/16 (post- mammal eradication).

		Coleoptera		Lepidoptera		Araneae	
		1986/87	2015/16	1986/87	2015/16	1986/87	2015/16
P1	Tall grass	25	1,909	0	0	151	124
P2	Tall grass	19	7	0	0	105	280
S1	*Stilbocarpa*	2,426	277	0	0	84	175
H1	Herbfield	8	6	0	0	27	123
H2	Herbfield	2	1	0	0	42	191
F1	Feldmark	4	3	0	0	8	120
F2	Feldmark	4	0	1	4	28	127

were collected principally via pitfall traps, most likely due their existence in soil or at the soil surface, particularly in moist sites. Their small size makes them unlikely to be detected through other trapping means. Cosmopolitan groups like Coleoptera, Collembola and Acarina were detected by all trapping methods. For the Collembola, their ubiquity in many samples exemplified their abundance and diversity on the island, with 35 species recorded (Phillips, et al., 2017). They also occur in a variety of habitats, from soil-dwellers to canopy species (Greenslade, 2006; Terauds, et al., 2011). Similarly, the collection of predatory Staphylinidae coleopterans across all trapping methods suggests this group are wide-ranging across vegetation, possibly to maximise opportunities to encounter prey. Isopoda, Annelidae and Platyhelminthes were collected by all means except sweeping (with a few exceptions), in line with their cryptic habits under vegetation, close to the soil surface and in litter (Greenslade, 2006).

Knowledge of the target group is critical to inform the experimental design of trapping. For example, and perhaps counter-intuitively, sweeping did not yield high numbers of moths or flies. One reason may be that many resident flies on Macquarie Island are flightless and largely stay close to the ground (Greenslade 2006). Furthermore, the endemic moth *Eudonia mawsoni*, which is not nocturnal, is known to drop to the ground when dislodged from vegetation (i.e. by sweep nets) (Jackson, 1995), and often stays close to the ground, taking shelter in winds over 10 km/hr (Greenslade, 2006). Sweeping can only occur during low wind conditions, however winds are typically high on the island (Pendlebury & Barnes-Keoghan, 2007), dispersing many taxa (both flightless and flying) (Hawes & Greenslade, 2013). The moth's flight is stimulated by rain, however sweeping is not possible during rains as wet vegetation renders the sweep net ineffective. Such background understanding of target taxa and the environment informs the design and interpretation of trapping surveys.

If the monitoring or management objectives focus on a particular group or species it is important to consider the varying effectiveness of trapping methods (Zou, et al., 2012). For example, mice on SOI prey mainly on invertebrates, especially spiders, moths, beetles, aphids, Orthoptera (e.g. crickets), snails and earthworms (Copson, 1986; Crafford & Sholtz, 1987; Rowe-Rowe, et al., 1989; Le Roux, et al., 2002; Jones, et al., 2003; Angel, et al., 2009; St Clair 2011; Russell, 2012). Copson (1986) identified that mice on Macquarie Island had a particular preference for spiders (67% of 108 mouse stomach contents), caterpillars of the endemic moth *E. mawsoni* and, to a lesser extent, other invertebrates such as beetles and dipteran larvae. Therefore, increases in these taxa following mouse eradication and the removal of predation pressure could be anticipated. Our preliminary comparisons provided some support for this hypothesis (see below). To measure the response of invertebrates preyed upon by mice on Macquarie Island, our results indicate that pitfall trapping is effective for spiders and beetles and is therefore the most efficient mechanism for assessing their recovery. Monitoring of moths will require greater consideration and ongoing effort, as they were not detected in high numbers by any trapping method during the 2015/16 season.

Comparisons to historic datasets are vital to detect responses to eradication. It is important to consider there may be different responses and recovery times in different species. Again, although our comparative analyses are only preliminary, they do show a higher abundance of spiders in pitfalls in 2015/16 compared with 1986/87 pitfalls, across all sites. There is a high likelihood that this is related to the eradication of mice, given spiders were a major prey item (Copson, 1986). However, beetle abundance did not change consistently between the two trapping events, with numbers trapped varying across sites and between years (Table 3). One possible reason is that Staphylinidae beetles (which comprised all of the beetles caught) can occur in dense numbers where rich detritus or rotting material is present on coastal terraces in vegetation (Greenslade 2006). As a result, they can be very abundant in an individual sample from one event, and then absent in others at the same site. Vegetation recovery is slow, and therefore, if beetle abundance is driven by vegetation, there will be a delay in beetle response to rabbit eradication.

For the moth *E. mawsoni*, despite anecdotal reports of increased abundance across the island, our preliminary data do not show this. The moth pitfall counts were similar in 1986/87 and 2015/16, with adults rarely trapped, which is consistent with other studies on Macquarie Island (Jackson, 1995; Potts, 1997; Stevens, et al., 2010; Hawes & Greenslade, 2013). The low number of adults in our data could be due to the timing of our sampling regime, i.e. our trapping effort was low in late December and early January – the time when adults are most abundant and active (Watson, 1967). Davies & Melbourne (1999) captured many adults using yellow pan traps. With this knowledge, we have added this method to our trapping regime for future seasons of the invertebrate monitoring project to identify change. We also extended our future sampling to occur later in the summer, between January–March, to identify seasonal changes in species, improve likelihood of encountering different species, and improve chances of detecting different life stages in species (such as the moths and moth larvae). Species life history must be considered when designing trapping to inform responses to management.

Terrestrial invertebrate communities that are dependent on or restricted to specific vegetation or habitat types are hypothesised to be most likely to be impacted by rabbit grazing on Macquarie Island (Parks and Wildlife, 2009). Vegetation has undergone considerable changes between 1986/87 and 2015/16 (Copson & Whinam, 1998; Bergstrom, et al., 2009; Shaw, et al., 2011). Our preliminary results highlight the potential utility of historical data, when combined with targeted and appropriate sampling techniques, to explore the relationship between vegetation and invertebrates. Overall, there appears to be an increase in richness and diversity from 1986/87 to 2015/16. During the period of the initial sampling in 1986/87, rabbit numbers and the commensurate vegetation damage were relatively high (Terauds, et al., 2014), which may explain the low numbers of vegetation-dependant invertebrates. Herbfields were favoured by rabbits and heavily impacted by grazing (Scott, 1988; Selkirk, et al., 1990; Copson & Whinam, 1998). Herbfield invertebrate communities were particularly low in richness and diversity in 1986/87. Feldmark communities vary little between 1986/87 and 2015/16, most likely as rabbit impacts were low in this high-altitude vegetation group (Copson & Whinam, 1998).

Another important consideration of these comparisons is that the historical survey data we accessed were generally based on higher taxonomic groupings, which may impact our ability to detect subtle changes in invertebrate communities over time (e.g. Grimbacher, et al., 2008). Higher taxonomic groupings, unsurprisingly, do not necessarily reflect the finer details of invertebrate community assemblages, nor nuanced changes in their structure and responses (Grimbacher, et al., 2008; Driessen & Kirkpatrick, 2017) and may aggregate species that have different ecological or functional traits and responses to disturbance (Lenat & Resh, 2001; Heino & Soininen, 2007; Schipper, et al., 2010). A limitation of using historical datasets is that often clarification and further detail is simply not available.

Identification of specimens to fine taxonomic resolution, such as to species level, takes considerable time. Although we focussed on higher-level taxa here, our forthcoming analyses at finer taxonomic resolution, (such as to family, genus and for some groups, species) will provide further insights on trapping efficacy, survey design and most importantly, invertebrate community changes over time. However, there is also good evidence to suggest that higher-level identification can be appropriate surrogates for species and effective for detecting major disturbance events on invertebrate community structure, particularly where there are significant environmental perturbations or gradients (Driessen & Włodarska-Kowalczuk & Kędra, 2007; Bevilacqua, et al., 2012; Kirkpatrick, 2017). Sorting samples to higher taxonomic resolution is often more practical and cost-effective as it requires less training and maximises the potential for swift sample processing (Driessen & Kirkpatrick, 2017). The selected taxonomic resolution must be balanced between available resources and the value of results – the decision ultimately depending on the study objectives (Driessen & Kirkpatrick, 2017). It is fundamentally important to decide at the outset which invertebrate taxa, (if not all of them), are the focus of the monitoring, and what taxonomic resolution will best meet the objectives of the monitoring. For example, the varying ranges and habitat associations of the five Staphylinidae beetle species on Macquarie Island are not represented when grouped to the Coleoptera order in our data. A second example is the ubiquitous and numerous Collembola (springtails), that when aggregated to Order, fail to highlight the very different species trapped by each medium, such as those that were trapped via sweeping which inhabit the canopy, foliage and flowers in vegetation, and those edaphic groups trapped via pitfalls that either inhabit the soil or live close to the surface. Such details are important to our Macquarie Island monitoring objectives as we assess invertebrate communities in recovering vegetation.

The availability of historical data greatly enhances the power of long-term effective monitoring. In this instance, considerable time was invested in tracking down historical datasets and their metadata. Our future work will include in-depth analysis of contemporary survey results in relation to a broader suite of historical data. Whether historical data are available at the outset or not, establishing a baseline from which to measure changes into the future is critical for long-term monitoring, for making informed management decisions, and assessing management success. Our preliminary results demonstrate that invertebrate monitoring in a post-vertebrate eradication ecosystem can yield important and promising results. Effective monitoring for invertebrates also leads to improved surveillance for non-native species arrivals and potential non-native species impacts. Our future work includes the collection of two additional years of invertebrate surveys (2016/17 and 2018) across Macquarie Island and the establishment of four additional invertebrate monitoring sites to improve island and vegetation community coverage. We will also employ additional trapping methods (vegetation beating and yellow-pan trapping), and use Berlese funnels in the 2016/17 and 2018 surveys for more efficient litter processing. These improvements combined, will further develop baseline knowledge of invertebrate communities on Macquarie Island and inform future monitoring. This work will provide a comprehensive snapshot of ecosystem function and recovery following vertebrate eradication. We will use these results to develop and propose an efficient means of invertebrate monitoring using specific taxa or groups as biological indicators of broader ecosystem changes, to enable robust and efficient monitoring into the future.

ACKNOWLEDGEMENTS

We thank Michael Driessen for input to research design and analyses. We thank Jasmine Lee for helpful comments on the draft manuscript. We are grateful for the helpful advice and comments given by two anonymous reviewers. We thank Kimberley Mitchell, Marcus Salton, George Brettingham-Moore, Rowena Hannaford, Jacqueline Comery and Penelope Pascoe for their assistance in the field with invertebrate trapping. We thank the Australian Antarctic Division for logistic support and the Tasmanian Parks and Wildlife Service for granting access to the island and permits to collect invertebrates (this work under Scientific Permit FA15234). This study was supported by funding from the Australian Government's National Environmental Science Programme through the Threatened Species Recovery Hub and the Australian Antarctic Science program (AAS 4305).

REFERENCES

Angel, A., Wanless, R.M. and Cooper, J. (2009). 'Review of impacts of the introduced house mouse on islands in the Southern Ocean: Are mice equivalent to rats?' *Biological Invasions* 11(7): 1743–1754.

Bergstrom, D.M., Lucieer, A., Kiefer, K., Wasley, J., Belbin, L., Pedersen, T.K. and Chown, S.L. (2009). 'Indirect effects of invasive species removal devastate world heritage island'. *Journal of Applied Ecology* 46(1): 73–81.

Bevilacqua, S., Terlizzi, A., Claudet, J., Fraschetti, S. and Boero, F. (2012). 'Taxonomic relatedness does not matter for species surrogacy in the assessment of community responses to environmental drivers'. *Journal of Applied Ecology* 49(2): 357–366.

Brennan, K.E., Majer, J.D. and Reygaert, N. (1999). 'Determination of an optimal pitfall trap size for sampling spiders in a Western Australian jarrah forest'. *Journal of Insect Conservation* 3(4): 297–307.

Chapuis, J.-L., Frenot, Y. and Lebouvier, M. (2004). 'Recovery of native plant communities after eradication of rabbits from the subantarctic Kerguelen Islands, and influence of climate change'. *Biological Conservation* 117(2): 167–179.

Chown, S.L. and Smith, V.R. (1993). 'Climate change and the short-term impact of feral house mice at the sub-Antarctic Prince Edward Islands'. *Oecologia* 96(4): 508–516.

Chown, S.L., Lee, J.E. and Shaw, J.D. (2008). 'Conservation of Southern Ocean islands: Invertebrates as exemplars'. *Journal of Insect Conservation* 12(3): 277–291.

Copson, G. R. (1986). 'The diet of the introduced rodents *Mus musculus* L. and *Rattus rattus* L on sub-Antarctic Macquarie Island'. *Wildlife Research* 13(3): 441–445.

Copson, G. and Whinam, J. (1998). 'Response of vegetation on subantarctic Macquarie Island to reduced rabbit grazing'. *Australian Journal of Botany* 46(1): 15–24.

Copson, G. and Whinam, J. (2001). 'Review of ecological restoration programme on subantarctic Macquarie Island: Pest management progress and future directions'. *Ecological Management and Restoration* 2(2): 129–138.

Courchamp, F., Chapuis, J.-L. and Pascal, M. (2003). 'Mammal invaders on islands: Impact, control and control impact'. *Biological Reviews* 78(03): 347–383.

Crafford, J.E. and Scholtz, C.H. (1987). 'Quantitative differences between the insect faunas of sub-antarctic Marion and Prince Edward islands: A result of human intervention?'. *Biological Conservation* 40(4): 255–262.

Davies, K.F. and Melbourne, B.A. (1999). 'Statistical models of invertebrate distribution on Macquarie Island: A tool to assess climate change and local human impacts'. *Polar Biology* 21: 240–250.

Driessen, M.M. and Kirkpatrick, J.B. (2017). 'Higher taxa can be effective surrogates for species-level data in detecting changes in invertebrate assemblage structure due to disturbance: a case study using a broad range of orders'. *Austral Entomology*: doi.org/10.1111/aen.12315.

Frenot, Y., Chown, S.L., Whinam, J., Selkirk, P.M., Convey, P., Skotnicki, M. and Bergstrom, D.M. (2005). 'Biological invasions in the Antarctic: Extent, impacts and implications'. *Biological Reviews* 80(1): 45–72.

Gerlach, J., Samways, M. and Pryke, J. (2013). 'Terrestrial invertebrates as bioindicators: An overview of available taxonomic groups'. *Journal of Insect Conservation* 17: 831–850.

Greenslade, P. (1987). Report on invertebrate fieldwork Macquarie Island, December 1986 – January 1987. Unpublished report. Tasmania, Australia: Department of Primary Industry, Parks, Water and the Environment.

Greenslade, P. (2006). *The Invertebrates of Macquarie Island.* Kingston: Australian Antarctic Division.

Grimbacher, P.S., Catterall, C.P. and Kitching, R.L. (2008). 'Detecting the effects of environmental change above the species level with beetles in a fragmented tropical rainforest landscape'. *Ecological Entomology* 33(1): 66–79.

Hawes, T.C. and Greenslade, P. (2013). 'The aerial invertebrate fauna of subantarctic Macquarie Island'. *Journal of Biogeography* 40(8): 1501–1511.

Heino, J. and Soininen, J. (2007). 'Are higher taxa adequate surrogates for species-level assemblage patterns and species richness in stream organisms?'. *Biological Conservation* 137(1): 78–89.

Hutcheson, J., Walsh, P. and Given, D.R. (1999). 'Potential Value of Indicator Species for Conservation and Management of New Zealand Terrestrial Communities'. *Science for Conservation* 109*.* Wellington, New Zealand: Department of Conservation.

Jackson, J. (1995). 'Notes on the biology of *Eudonia mawsoni* (Lepidoptera: Pyralidae) on subantarctic Macquarie Island'. *Polar Biology* 15(4): 289–294.

Jones, A.G., Chown, S.L., Webb, T.J. and Gaston, K.J. (2003). 'The free-living pterygote insects of Gough Island, South Atlantic Ocean'. *Systematics and Biodiversity* 1(2): 213–273.

Kremen, C., Colwell, R.K., Erwin, T.L., Murphy, D.D., Noss, R.F. and Sanjayan, M.A. (1993). 'Terrestrial arthropod assemblages: Their use in conservation planning'. *Conservation biology* 7(4): 796–808.

Le Roux, V., Chapuis, J.-L., Frenot, Y. and Vernon, P. (2002). 'Diet of the house mouse (*Mus musculus*) on Guillou Island, Kerguelen archipelago, Subantarctic'. *Polar Biology* 25(1):49–57.

Lenat, D.R. and Resh, V.H. (2001). 'Taxonomy and stream ecology—the benefits of genus-and species-level identifications'. *Journal of the North American Benthological Society* 20(2): 287–298.

McGeoch, M.A., Sithole, H., Samways, M.J., Simaika, J.P., Pryke, J.S., Picker, M., Uys, C., Armstrong, A.J., Dippenaar-Schoeman, A.S. and Engelbrecht, I.A. (2011). 'Conservation and monitoring of invertebrates in terrestrial protected areas'. *Koedoe* 53(2): 131–143.

McGeoch, M.A., Shaw, J.D., Terauds, A., Lee, J.E. and Chown, S.L. (2015). 'Monitoring biological invasion across the broader Antarctic: A baseline and indicator framework'. *Global Environmental Change* 32: 108–125.

Parks and Wildlife. (2009). *Macquarie Island Pest Eradication Plan. Part C: Environmental Impact Statement*. Hobart, Tasmania: Department of Primary Industries, Parks, Water and the Environment.

Pendlebury, S. and Barnes-Keoghan, I. (2007) 'Climate and climate change in the sub-Antarctic'. *Papers and Proceedings of the Royal Society of Tasmania* 141(1): 67–81.

Phillips, L., Janion-Scheepers, C., Houghton, M., Terauds, A., Potapov, M. and Chown, S.L. (2017). 'Range expansion of two invasive springtails on sub-Antarctic Macquarie Island'. *Polar Biology*: 1–6.

Potts, T. (1997). 'Monitoring Invertebrate Migration and Dispersal on Subantarctic Macquarie Island'. BSc Honours Thesis. Hobart: University of Tasmania, Institute of Antarctic and Southern Ocean Studies (IASOS).

Robinson, S.A. and Copson, G.R. (2014). 'Eradication of cats (*Felis catus*) from subantarctic Macquarie Island'. *Ecological Management and Restoration* 15(1): 34–40.

Rowe-Rowe, D., Green, B. and Crafford, J. (1989). 'Estimated impact of feral house mice on sub-Antarctic invertebrates at Marion Island'. *Polar Biology* 9(7): 457–460.

Russell, J.C. (2012). 'Spatio-temporal patterns of introduced mice and invertebrates on Antipodes Island'. *Polar Biology* 35(8):1187–1195.

Samways, M.J., McGeoch, M.A. and New, T.R. (2010). *Insect Conservation: A Handbook of Approaches and Methods*. Oxford: Oxford University Press.

Schipper, A.M., Lotterman, K., Geertsma, M., Leuven, R.S. and Hendriks, A.J. (2010). 'Using datasets of different taxonomic detail to assess the influence of floodplain characteristics on terrestrial arthropod assemblages'. *Biodiversity and Conservation* 19(7): 2087–2110.

Scott, J. (1988) 'Rabbit distribution history and related land disturbance, Macquarie Island'. *Papers and Proceedings of the Royal Society of Tasmania* 122(1): 255–266.

Scott, J. and Kirkpatrick, J. (2008). 'Rabbits, landslips and vegetation change on the coastal slopes of subantarctic Macquarie Island, 1980–2007: implications for management'. *Polar Biology* 31(4): 409–419.

Scott, J.J. and Kirkpatrick, J.B. (2013). 'Changes in the cover of plant species associated with climate change and grazing pressure on the Macquarie Island coastal slopes, 1980–2009'. *Polar Biology* 36(1): 127–136.

Selkirk, P., Seppelt, R. and Selkirk, D. (1990). *Subantarctic Macquarie Island: Environment and Biology*. Cambridge: Cambridge University Press.

Shaw, J., Terauds, A. and Bergstrom, D. (2011). 'Rapid commencement of ecosystem recovery following aerial baiting on sub-Antarctic Macquarie Island'. *Ecological Management and Restoration* 12(3): 241–244.

Smith, V.R. and Steenkamp, M. (1990). 'Climatic change and its ecological implications at a subantarctic island'. *Oecologia* 85: 14–24.

Smith, V.R. and Steenkamp, M. (1992). 'Soil macrofauna and nitrogen on a sub-Antarctic island'. *Oecologia* 92(2): 201–206.

Smith, V.R. (2007). 'Introduced slugs and indigenous caterpillars as facilitators of carbon and nutrient mineralisation on a sub-Antarctic island'. *Soil Biology and Biochemistry* 39: 709–713.

Smith, V.R. (2008). 'Energy flow and nutrient cycling in the Marion Island terrestrial ecosystem: 30 years on'. *Polar Record* 44(3): 211–226.

Springer, K. (2016). 'Methodology and challenges of a complex multi-species eradication in the sub-Antarctic and immediate effects of invasive species removal'. *New Zealand Journal of Ecology* 40(2): 273–278.

St Clair, J.J.H. (2011). 'The impacts of invasive rodents on island invertebrates'. *Biological Conservation* 144(1): 68–81.

St Clair, J.J.H., Poncet, S., Sheehan, D.K., Székely, T. and Hilton, G.M. (2011). 'Responses of an island endemic invertebrate to rodent invasion and eradication'. *Animal Conservation* 14(1): 66–73.

Stevens, M., Hudson, P., Greenslade, P. and Potter, M. (2010). *Report on invertebrate monitoring of long term field sites on Macquarie Island 2009/2010*. Unpublished report. South Australia: South Australia Museum.

Terauds, A., Chown, S.L. and Bergstrom, D.M. (2011). 'Spatial scale and species identity influence the indigenous–alien diversity relationship in springtails'. *Ecology* 92(7): 1436–1447.

Terauds, A., Doube, J., McKinlay, J. and Springer, K. (2014). 'Using long-term population trends of an invasive herbivore to quantify the impact of management actions in the sub-Antarctic'. *Polar Biology* 37(6): 833–843.

Thoresen, J.J., Towns, D., Leuzinger, S., Durrett, M., Mulder, C.P. and Wardle, D.A. (2017). 'Invasive rodents have multiple indirect effects on seabird island invertebrate food web structure'. *Ecological Applications* 27(4): 1190–1198.

Towns, D.R., Wardle, D.A., Mulder, C.P., Yeates, G.W., Fitzgerald, B.M., Parrish, R.G., Bellingham, P.J. and Bonner, K.I. (2009). 'Predation of seabirds by invasive rats: Multiple indirect consequences for invertebrate communities'. *Oikos* 118(3): 420–430.

Treasure, A.M., Chown, S.L. and Diez, J. (2014). 'Antagonistic effects of biological invasion and temperature change on body size of island ectotherms'. *Diversity and Distributions* 20(2): 202–213.

Vogel, M., Remmert, H. and Lewis-Smith, R.I. (1984). 'Introduced reindeer and their effects on the vegetation and the epigeic invertebrate fauna of South Georgia (subantarctic)'. *Oecologia* 62(1): 102–109.

Ward, D.F., New, T.R. and Yen, A.L. (2001). 'Effects of pitfall trap spacing on the abundance, richness and composition of invertebrate catches'. *Journal of Insect Conservation* 5(1): 47–53.

Watson, K.C. (1967). 'The terrestrial arthropoda of Macquarie Island'. *ANARE Scientific Reports, Series B (1) Zoology* 99: 1–90.

Whinam, J., Fitzgerald, N., Visoiu, M. and Copson, G. (2014). 'Thirty years of vegetation dynamics in response to a fluctuating rabbit population on sub-Antarctic Macquarie Island'. *Ecological Management and Restoration* 15(1): 41–51.

Włodarska-Kowalczuk, M. and Kędra, M. (2007). 'Surrogacy in natural patterns of benthic distribution and diversity: selected taxa versus lower taxonomic resolution'. *Marine Ecology Progress Series* 351: 53–63.

Woodcock, B.A. (2005). 'Pitfall Trapping in Ecological Studies'. In: S. Leather (ed.) *Insect Sampling in Forest Ecosystems*, pp. 37–57. Oxford: Blackwell.

Work, T.T., Buddle, C.M., Korinus, L.M. and Spence, J.R. (2002). 'Pitfall trap size and capture of three taxa of litter-dwelling arthropods: Implications for biodiversity studies'. *Environmental Entomology* 31(3): 438–448.

Zou, Y., Jinchao, F., Dayuan, X., Weiguo, S. and Axmacher, J.C. (2012). 'A comparison of terrestrial arthropod sampling methods'. *Journal of Resources and Ecology* 3(2): 174–182.