special issue:
Feathers to Fur
The ecological transformation of Aotearoa/New Zealand

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

Peter J. Bellingham[1]*, David R. Towns[2], Ewen K. Cameron[3], Joe J. Davis[4], David A. Wardle[1, 5], Janet M. Wilmshurst[1] and Christa P.H. Mulder[6]
[1]Landcare Research, PO Box 40, Lincoln 7640, New Zealand
[2]Department of Conservation, Private Bag 68-908, Auckland, New Zealand
[3]Auckland Museum, Private Bag 92018, Auckland, New Zealand
[4]Ngāti Hei Trust, PO Box 250, Whitianga, New Zealand
[5]Department of Forest Vegetation Ecology, Swedish University of Agricultural Sciences, S 901 83 Umeå, Sweden
[6]Department of Biology and Wildlife & Institute of Arctic Biology, University of Alaska Fairbanks, AK 99775, USA
*Author for correspondence (Email: bellinghamp@landcareresearch.co.nz)

Published online: 6 October 2009

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 primaeval 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.

This special issue reviews the current status of New Zealand ecology, updating the 1989 *Moas Mammals and Climate* special issue (NZJ Ecol 12 supplement). Both issues are available at www.newzealandecology.org.nz/nzje/.

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

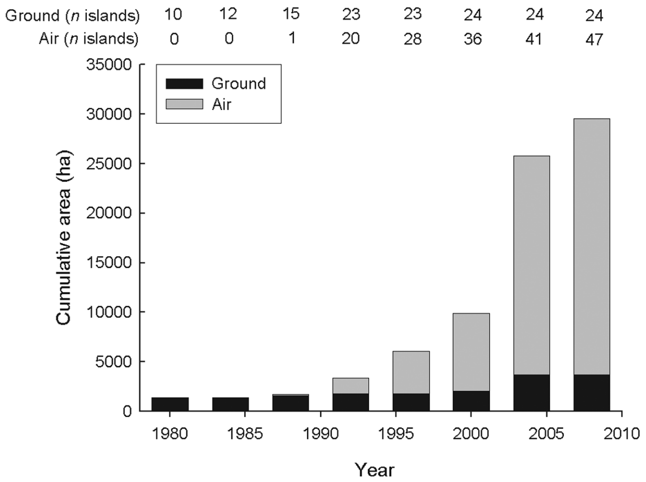

Figure 1. Cumulative area of islands around New Zealand cleared of all non-native mammals since 1980 using ground-based control methods alone (dark bars) or aerial sowing of poison baits (open bars), which for some islands may have been in combination with ground based methods (D.R. Towns, unpublished 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 crake *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-estabished 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 *Placostylus ambagiosus*, snail *Amborhytida taranagaensis*, ghost crab *Ocypode kuhlii*, land crab *Geograpsus grayi*, Cook Strait giant wētā *Deinacrida rugosa**, Mahoenui wētā *D. mahoenui**, Auckland tree wētā *Hemidiena thoracica**, Wellington tree wētā *H. crassidens**, Mercury Island tusked wētā *Motuweta isolata**, large darkling beetle *Mimopeus opaculus**, honeydew scale *Coelostomidia zealandica*, flax weevil *Anagotus fairburni**, Turbott's karo weevil *A. turbotti**, knobbled weevil *Hadramphus stilbocarpae**, speargrass weevil *Lyperobius huttoni**
Amphibians	Hamilton's frog *Leiopelma hamiltoni**, Maud Island frog *L. pakeka**
Reptiles	Northern tuatara *Sphendon p. punctatus*(*), Cook Strait tuatara *S. p.* subsp. unnamed*, Brothers Island tuatara *S. guntheri** , common gecko *Hoplodactylus maculatus*, Duvaucel's gecko *H. duvaucelii* (*), forest gecko *H. granulatus**, Matapia gecko *H.* sp*, Pacific gecko *H. pacificus* (*), Marlborough green gecko *Naultinus manukanus**, Wellington green gecko *N. elegans punctatus**, copper skink *Cyclodina aenea*, marbled skink *C. oliveri**, McGregor's skink *C. macgregori* (*), Mokohinau skink *C. townsi* (*), ornate skink *C. ornata* (*), robust skink *C. alani**, Whitaker's skink *C. whitakeri**, brown skink *Oligosoma zelandicum*, chevron skink *O. homalonotum*, common skink *O. nigriplantare polychroma**, Fiordland skink *Oligosoma acrinasum*, moko skink *O. moco*, shore skink *O. smithi* (*), speckled skink *O. infrapunctatum**, spotted skink *O. lineoocellatum* (*), Suter's skink *O. suteri*(*)
Terrestrial birds	Brown kiwi *Apteryx mantelli**, little spotted kiwi *A. owenii**, brown teal *Anas aucklandica chlorotis**, Campbell Island teal *A. a. aucklandica**, spotless crake *Porzana tabuensis plumbea*, North Island weka *Gallirallus australis greyi**, New Zealand dotterel *Charadrius obscurus*, Campbell Island snipe *Coenocorypha aucklandica* subsp. undescribed, sooty tern *Sterna fuscata*, kākāpō *Strigops habroptilus**, Kermadec parakeet *Cyanoramphus novaezelandiae cyanurus*, red-crowned parakeet *C. n. novaezelandiae**, yellow-crowned parakeet *C. auriceps**, kingfisher *Halcyon sancta vagans*, rock wren *Xenicus gilviventris**, pipit Campbell Island *Anthus novaeseelandiae aucklandicus*, North Island fernbird *Bowdleria punctata vealae**, Stewart Island fernbird *B. p. stewartiana**, whitehead *Mohoua albicilla**, mohua *M. ochrocephala**, North Island robin *Petroica australis longipes**, South Island robin *P. a. australis**, hihi *Notiomystis cincta**, North Island tieke *Philesturnus carunculatus rufusater**, South Island tieke *P. c. carunculatus**, kōkako *Calleas cinerea wilsoni**
Seabirds	Grey-faced petrel *Pterodroma macroptera gouldi*, Kermadec petrel *P. n. neglecta*, Cook's petrel *P. cookii*, Pycroft's petrel *P. pycrofti* (*), fairy prion *Pachyptila turtur**, black petrel *Procellaria parkinsoni* (*), white-chinned petrel *P. aequinoctialis*, wedge-tailed shearwater *Puffinus pacificus*, fluttering sheawater *P. gavia*, little shearwater *P. assimilis haurakiensis*, grey-backed storm petrel *Garrodia nereis*, diving petrel *Pelecanoides urinatrix*, red-tailed tropic bird *Phaeton rubricauda reseotincta*

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 (Towns 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 (Towns 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 (Towns & Ferreira 2001, D.R. Towns 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 (Towns & 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 kākāriki (*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 (*Anplolepis 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 *Vespula* 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 ^{14}C 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 ^{14}C 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 spelaeus*) 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 $\delta^{15}N$ in soils was 23% lower (Fukami et al. 2006). Reduced soil resources on rat-invaded islands has resulted in reduced microbial CO_2 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 are kiore are present (Campbell & Atkinson 2002). However experiments are needed to determine

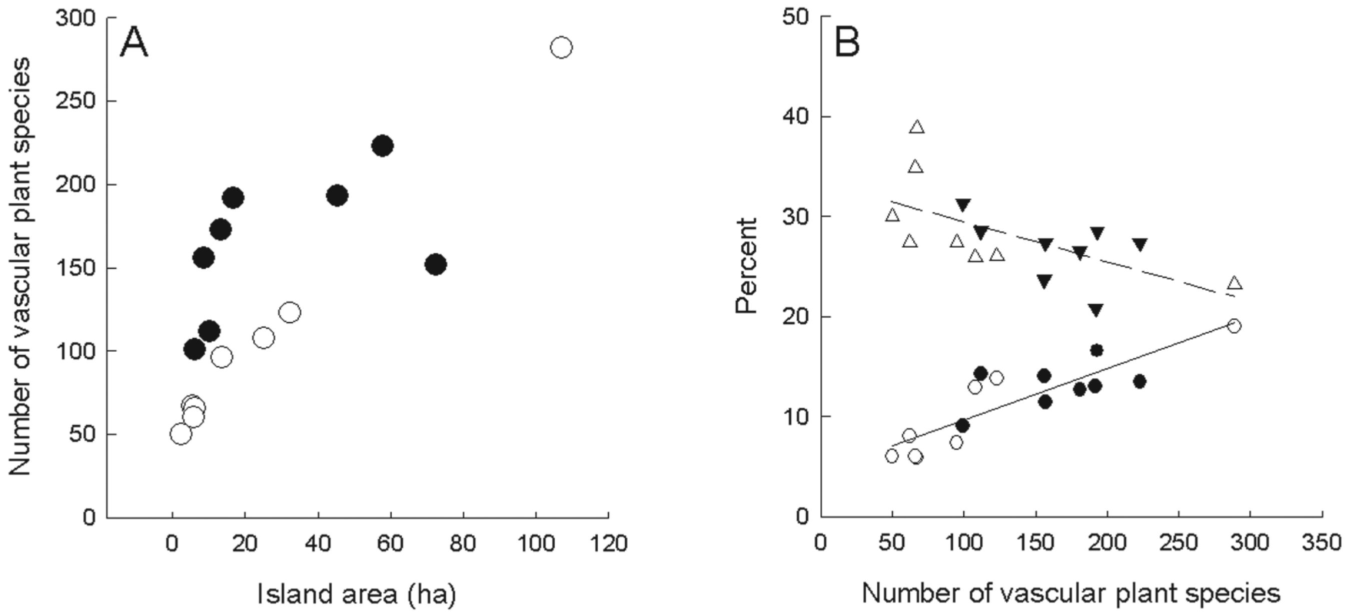

Figure 2. A. Relationship between island area and the number of vascular plant species on 16 islands in warm temperate northern New Zealand; open symbols = islands never invaded by rats; filled symbols = islands invaded by ship or Norway rats. B. Relationships between the number of vascular plant species 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.

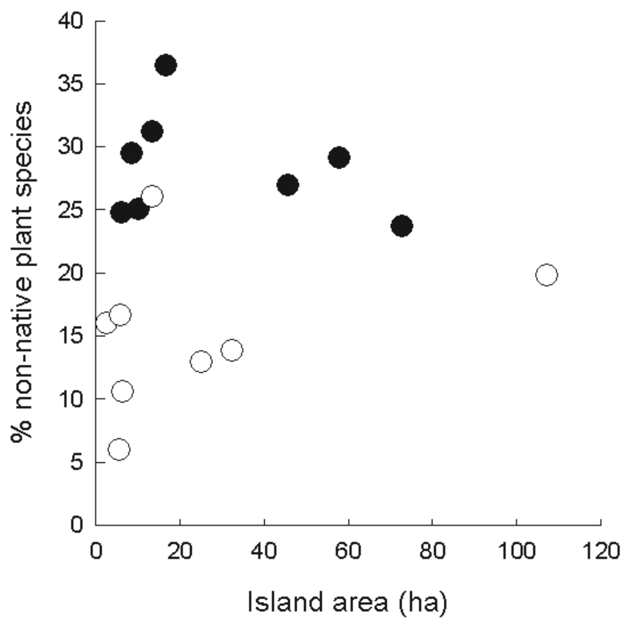

Figure 3. Relationship between island area and the percentage of vascular plant species that are non-native on 16 islands in warm temperate northern New Zealand; open symbols = islands never invaded by rats; filled symbols = islands invaded by ship or Norway rats. Data sources listed in Appendix 4.

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 ± 1.49% of the total flora vs. uninvaded islands, 15.2 ± 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 × 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 is 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.

Acknowledgements

We received funding from the Marsden Fund, the Foundation for Research, Science, and Technology (Sustaining and Restoring Biodiversity Outcome-based Investment), the US National Science Foundation and the Department of Conservation. The contribution by DRT includes part of a plenary address to the "Biodiversity Extinction Crisis" Conference, Sydney, Australia (July 2007), supported by the Australasian branch of the Society for Conservation Biology. For access to islands we thank the Ngātiwai Trust; Ngāti Rehua, Ngāti Manuhiri; Ngāti Paoa; Ngāti Hei; Ngāti Hako; Ngāti Puu; Ngāmotuaroha Islands Trust, Rumāhua (Aldermen) Islands Trust, the Neureuter family, Oho Nicolls, and the Department of Conservation. We thank Phil Lyver and John Parkes for input. Dave Kelly, Bill Lee and an anonymous referee provided valuable review comments. We thank the Department of Conservation and Landcare Research for sponsoring the publication of this special journal issue.

References

Abbott KL, Green PT 2007. Collapse of an ant–scale mutualism in a rainforest on Christmas Island. Oikos 116: 1238–1246.

Adern SL, Lambert DM 1997. Is the black robin in genetic peril? Molecular Ecology 6: 21–28.

Allen RB, Lee WG, Rance BD 1994. Regeneration in indigenous forest after eradication of Norway rats, Breaksea Island, New Zealand. New Zealand Journal of Botany 32: 429–439.

Alley MR, Castro I, Hunter JEB 1999. Aspergillosis in hihi (*Notiomystis cincta*) on Mokoia Island. New Zealand Veterinary Journal 47: 88–91.

Amey J, Lord JM, de Lange P 2007. First record of a vascular plant from the Bounty Islands: *Lepidium oleraceum* (nau, Cook's scurvy grass) (Brassicaceae). New Zealand Journal of Botany 45: 87–90.

Anderson AJ 2005. Subpolar settlement in south Polynesia. Antiquity 79: 791–800.

Anderson S, Heiss-Dunlop S, Flohr J 2006. A moving feast: the influence of landscape context on bird-mediated seed rain into conservation areas. In: Preston C, Watts JH, Crossman ND eds. 15th Australian Weeds Conference Proceedings: managing weeds in a changing climate. Adelaide, Weed Management Society of South Australia Inc. Pp. 66–71.

Anyan S, Lang S 2008. The sweet smell of success. North and South 263: 68–78.

Armstrong DP, McLean IG 1995. New Zealand translocations: theory and practice. Pacific Conservation Biology 2: 95–102.

Armstrong, DP, Perrott JK 2000. An experiment testing whether conditions and survival are limited by food supply in a reintroduced hihi population. Conservation Biology 14: 1171–1181.

Armstrong DP, Castro I, Griffiths R 2007. Using adaptive management to determine requirements of re-introduced populations: the case of New Zealand hihi. Journal of Applied Ecology 44: 953–962.

Atkinson IAE 1964. The flora, vegetation, and soils of Middle and Green Island, Mercury Islands group. New Zealand Journal of Botany 2: 385–402.

Atkinson IAE 1988. Opportunities for ecological restoration. New Zealand Journal of Ecology 11: 1–12.

Atkinson IAE 1990. Ecological restoration on islands: prerequisites for success. In: Towns DR, Daugherty CH, Atkinson IAE eds. Ecological restoration of New Zealand islands. Wellington, Department of Conservation. Pp. 73–90.

Atkinson IAE 2004. Successional processes induced by fires on the northern offshore islands of New Zealand. New Zealand Journal of Ecology 28: 181–193.

Bailey AM, Sorensen JH 1962. Subantarctic Campbell Island. Proceedings of the Denver Museum of Natural History 10.

Ballance A 2007. Don Merton: the man who saved the black robin. Auckland, Reed.

Barker D, Carroll JWA, Edmonds HK, Fraser JR, Miskelly CM 2005. Discovery of a previously unknown *Coenocorypha* snipe in the Campbell Island group, New Zealand subantarctic. Notornis 52: 143–149.

Baylis GTS 1948. Vegetation on Great Island, Three Kings Group. Records of the Auckland Institute and Museum 3: 239–256.

Baylis GTS 1951. Incipient forest regeneration on Great Island, Three Kings Group. Records of the Auckland Institute and Museum 4: 103–109.

Baylis GTS 1986. Widened seed dispersal at the Three Kings Islands attributed to naturalised birds. In: Wright AE, Beever RE eds. The offshore islands of northern New Zealand. Department of Lands and Survey Information Series 16. Pp. 41–45.

Beggs J 2001. The ecological consequences of social wasps (*Vespula* spp.) invading an ecosystem that has an abundant carbohydrate resource. Biological Conservation 99: 17–28.

Beggs JR, Wardle DA 2006. Keystone species: competition for honeydew among exotic and indigenous species. In: Allen RB, Lee WG eds. Biological invasions in New Zealand. Berlin, Springer. Pp. 281–294.

Bell BD 1978. The Big South Cape islands rat irruption. In: Dingwall PR, Atkinson IAE, Hay C eds. The ecology and control of rodents in New Zealand nature reserves. Department of Lands and Survey Information Series 4. Pp. 33–40.

Bell M, Bell BD, Bell EA 2005. Translocation of fluttering shearwater (*Puffinus gavia*) chicks to create a new colony. Notornis 52: 11–15.

Brockie RE 1983. Starlings (*Sturnus vulgaris*) roosts and flightlines near Wellington. Notornis 30: 217–226.

Brook FJ 2000. Prehistoric predation of the landsnail *Placostylus ambagiosus* Suter (Stylommatophora: Bulimulidae), and evidence for the timing of establishment of rats in northernmost New Zealand. Journal of the Royal Society of New Zealand 30: 227–241.

Burrows CJ 2008. Genus *Pimelea* (Thymelaeaceae) in New Zealand 1. The taxonomic treatment of seven endemic, glabrous-leaved species. New Zealand Journal of Botany 46: 127–176.

Butler D, Merton D 1992. The black robin: saving the world's most endangered bird. Auckland, Oxford University Press.

Cameron EK 1984. Vascular plants of the three largest Chickens (Marotere) Islands: Lady Alice, Whatupuke, Coppermine: north-east New Zealand. Tane 30: 53–75.

Cameron EK 1990. Flora and vegetation of Middle Island, Mercury Islands Group, eastern Coromandel, northern New Zealand. Journal of the Royal Society of New

Zealand 20: 273–285.
Campbell DJ, Atkinson IAE 1999. Effects of kiore (*Rattus exulans* Peale) on recruitment of indigenous coastal trees on northern offshore islands of New Zealand. Journal of the Royal Society of New Zealand 29: 265–290.
Campbell DJ, Atkinson IAE 2002. Depression of tree recruitment by the Pacific rat (*Rattus exulans* Peale) on New Zealand's northern offshore islands. Biological Conservation 107: 19–35.
Caro T 2007. Behavior and conservation: a bridge too far? Trends in Ecology and Evolution 22: 394–400.
Castanet J, Newman DG, Saint Girons H 1988. Skeletochronological data on the growth, age, and population structure of the tuatara, *Sphenodon punctatus*, on Stephens and Lady Alice Islands, New Zealand. Herpetologica 44: 25–37.
Castro I, Alley JC, Empson RA, Minot EO 1994. Translocation of hihi or stitchbird *Notiomystis cincta* to Kapiti Island, New Zealand: transfer techniques and comparison of release strategies. In: Serena M ed. Reintroduction biology of Australian and New Zealand fauna. Chipping Norton, Surrey Beatty & Sons. Pp. 113–120.
Cheeseman TF 1888. Notes on the Three Kings Islands. Transactions and Proceedings of the New Zealand Institute 20: 141–150 and 429–439.
Clarkson BD, Clarkson BR 1994. Vegetation decline following recent eruptions on White Island (Whakaari), Bay of Plenty, New Zealand. New Zealand Journal of Botany 32: 21–36.
Clout MN, Hay JR 1989. The importance of birds as browsers, pollinators and seed dispersers in New Zealand forests. New Zealand Journal of Ecology 12 (supplement): 27–33.
Clout MN, Elliott GP, Robertson BC 2002. Effects of supplementary feeding on the offspring sex ratio of kakapo: A dilemma for the conservation of a polygynous parrot. Biological Conservation 107: 13–18.
Cochrane GR 1962. The Aldermen Islands: a remnant of primaeval New Zealand. Transactions of the Royal Society of New Zealand – Botany 1: 331–341.
Cockayne L 1904. A botanical excursion during midwinter to the southern islands of New Zealand. Transactions and Proceedings of the New Zealand Institute 36: 225–333.
Cockayne L 1905. Notes on a brief botanical visit to the Poor Knights Islands. Transactions and Proceedings of the New Zealand Institute 38: 351–360.
Coomes DA, Allen RB, Bentley WA, Burrows LE, Canham CD, Fagan L, Forsyth DM, Gaxiola-Alcantar A, Parfitt RL, Ruscoe WA, Wardle DA, Wilson DJ, Wright EF 2005. The hare, the tortoise and the crocodile: the ecology of angiosperm dominance, conifer persistence and fern filtering. Journal of Ecology 93: 918–935.
Courchamp F, Chapuis JL, Pascal M 2003. Mammal invaders on islands: impact, control and control impact. Biological Reviews 78: 347–383.
Court DJ, Hardacre AK, Lynch PA 1981. The vegetation of the Aldermen Islands: a reappraisal. Tane 27: 41–60.
Cowan PE 1992. The eradication of introduced Australian brushtail possums, *Trichosurus vulpecula*, from Kapiti Island, a New Zealand nature reserve. Biological Conservation 61: 217–226.
Cree A 1994. Low annual reproductive output in female reptiles from New Zealand. New Zealand Journal of Zoology 21: 351–372.
Daugherty CH, Towns DR, Cree A, Hay JM 1992. The roles of legal protection versus intervention in conserving New Zealand tuatara, *Sphenodon*. In: Willison JH, Bondrup-Nielson S, Drysdale C, Herman TB, Munro NWP, Pollock TL eds. Science and the management of protected areas. Amsterdam, Elsevier. Pp. 247–259.
Davidson J 1984. The prehistory of New Zealand. Auckland, Longman Paul.
de Lange PJ, Cameron EK 1999. The vascular flora of Aorangi Island, Poor Knights Islands, northern New Zealand. New Zealand Journal of Botany 37: 433–468.
Diamond JM, Veitch CR 1981. Extinctions and introductions in the New Zealand avifauna: cause and effect? Science 211: 499–501.
Donlan CJ, Wilcox C 2008. Integrating invasive mammal eradications and biodiversity offsets for fisheries bycatch: conservation opportunities and challenges for seabirds and sea turtles. Biological Invasions 10: 1053–1060.
Duncan RP, Forsyth DM 2006. Modelling population persistence on islands: mammal introductions in the New Zealand archipelago. Proceedings of the Royal Society B 273: 2969–2975.
Elliott GP, Eason DK, Jansen PW, Merton DV, Harper GA, Moorhouse RJ 2006. Productivity of kakapo (*Strigops habroptilus*) on offshore island refuges. Notornis 53: 138–142.
Ellis JC 2005. Marine birds on land: a review of plant biomass, species richness, and community composition in seabird colonies. Plant Ecology 181: 227–241.
Empson L, Flenley J, Sheppard P 2002. A dated pollen record of vegetation change on Mayor Island (Tuhua) throughout the last 3000 years. Global and Planetary Change 33: 329–337.
Erdtman G 1924. Studies in micro-palaeontology IV. Peat from the Chatham islands and the Otago District, New Zealand. Geologiska Föreningen; Stockholm Förhandlingar 46: 679–681.
Esler AE 1967. The vegetation of Kapiti Island. New Zealand Journal of Botany 5: 353–394.
Ferguson RN, Drake DR 1999. Influence of vegetation structure on spatial patterns of seed deposition by birds. New Zealand Journal of Botany 37: 671–677.
Fleming CA 1979. The geological history of New Zealand and its life. Auckland, Oxford University Press.
Fleming CA 1982. George Edward Lodge: the unpublished New Zealand bird paintings. Wellington, Nova Pacifica.
Freeman AND 1994. Landbirds recorded at the Chatham Islands, 1940 to December 1993. Notornis 41 (Supplement): 127–141.
Fukami T, Wardle DA, Bellingham PJ, Mulder CPH, Towns DR, Yeates GW, Bonner KI, Durrett MS, Grant-Hoffman MN, Williamson WM 2006. Above- and below-ground impacts of introduced predators in seabird-dominated island ecosystems. Ecology Letters 9: 1299–1307.
Galbreath R, Brown D 2004. The tale of the lighthouse-keeper's cat: discovery and extinction of the Stephens Island wren (*Traversia lyalli*). Notornis 51: 193–200.
Gibbs G 2006. Ghosts of Gondwana. Nelson, Craig Potton Publishing.
Gillham ME 1956a. Ecology of the Pembrokeshire Islands. V. Manuring by the colonial seabirds and mammals, with a note on seed distribution by gulls. Journal of Ecology 44: 429–454.
Gillham ME 1956b. Ecology of the Pembrokeshire Islands. IV. Effects of treading and burrowing by birds and mammals. Journal of Ecology 44: 51–82.

Gillham ME 1960a. Vegetation of New Zealand shag colonies. Transactions of the Royal Society of New Zealand 88: 363–380.

Gillham ME 1960b. Plant communities of the Mokohinau Islands, northern New Zealand. Transactions of the Royal Society of New Zealand 88: 79–98.

Gillham ME 1960c. Destruction of indigenous heath vegetation in Victorian sea-bird colonies. Australian Journal of Botany 8: 277–317.

Gillham ME 1960d. Vegetation of Little Brother Island, Cook Strait, in relation to spray-bearing winds, soil salinity, and pH. Transactions of the Royal Society of New Zealand 88: 405–424.

Haines L, Julian A, Wilcox MD 2007. Vegetation patterns. In: Wilcox MD ed. Natural history of Rangitoto Island. Auckland, Auckland Botanical Society. Pp. 41–58.

Hamilton WM (ed.) 1961. Little Barrier Island (Hauturu). New Zealand Department of Scientific and Industrial Research Bulletin No. 137. Wellington, Government Printer.

Hamilton WM, Atkinson IA 1961. Vegetation. In: Hamilton WM ed. Little Barrier Island (Hauturu). New Zealand Department of Scientific and Industrial Research Bulletin No. 137. Wellington, Government Printer. Pp. 87–121.

Hamilton WM, Baumgart IL (eds.) 1959. White Island. New Zealand Department of Scientific and Industrial Research Bulletin No. 127. Wellington, Government Printer.

Harper PC 1983. Biology of the Buller's shearwater (*Puffinus bulleri*) at the Poor Knights Islands, New Zealand. Notornis 30: 299–318.

Harper GA 2007. Detecting predation of a burrow-nesting seabird by two introduced predators, using stable isotopes, dietary analysis and experimental removals. Wildlife Research 34: 443–453.

Harris RJ 2002. Potential impact of the Argentine ant (*Linepithema humile*) in New Zealand and options for its control. Science for Conservation 196. Wellington, Department of Conservation.

Harris RJ, Etheridge ND 2001. Comparison of baits containing fiprinol and sulfluramid for the control of *Vespula* wasps. New Zealand Journal of Zoology 28: 39–48.

Hawke DJ, Holdaway RN, Causer JE, Ogden S 1999. Soil indicators of pre-European seabird breeding in New Zealand at sites identified by predator deposits. Australian Journal of Soil Research 37: 103–113.

Hawke DJ, Newman J 2007. Carbon-13 and nitrogen-15 enrichment in coastal forest foliage from nutrient-poor and seabird-enriched sites in southern New Zealand. New Zealand Journal of Botany 45: 309–315.

Hawke D, Newman J, Moller H, Wixon J 2003. A possible early muttonbirder's fire on Poutama, a Rakiura titi island, New Zealand. Journal of the Royal Society of New Zealand 33: 497–507.

Hay CH, Warham J, Fineran BA 2004. The vegetation of The Snares, islands south of New Zealand, mapped and discussed. New Zealand Journal of Botany 42: 861–871.

Heather BD, Robertson HA 1996. The field guide to the birds of New Zealand. Auckland, Viking.

Horrocks M, Deng Y, Ogden J, Alloway BV, Nichol SL, Sutton DG 2001. High spatial resolution of pollen and charcoal in relation to the *c.* 600-year-BP Kaharoa Tephra: implications for Polynesian settlement of Great Barrier Island, northern New Zealand. Journal of Archaeological Science 28: 153–168.

Hunt DS, Williams M 2000. Restoration plan for Tuhua (Mayor Island). Rotorua, Department of Conservation.

Imber MJ 1973. The food of grey-faced petrels (*Pterodroma macroptera gouldi* (Hutton)), with special reference to diurnal vertical migration of their prey. Journal of Animal Ecology 42: 645–662.

Imber MJ, West JA, Cooper WJ 2003a. Breeding biology of Cook's petrel (*Pterodroma cookii*) on Hauturu (Little Barrier Island) and Whenua Hou (Codfish Island). Notornis 50: 221–230.

Imber MJ, McFadden I, Bell EA, Scofield RP 2003b. Post-fledging migration, age of first return and recruitment, and results of inter-colony translocations of black petrels (*Procellaria parkinsoni*). Notornis 50: 183–190.

Innes J, Kelly D, Overton J, Gillies C 2010. Predation and other factors currently limiting New Zealand forest birds. New Zealand Journal of Ecology 34: 86–114.

Jackson AL, Bearhop S, Thompson DR 2005. Shape can influence the rate of colony fragmentation in ground nesting seabirds. Oikos 111: 473–478.

Jamieson IG, Ryan CJ 2000. Increased egg infertility associated with translocating inbred takahe (*Porphyrio hochstetteri*) to island refuges in New Zealand. Biological Conservation 94: 107–114.

Jamieson IG, Roy MS, Lettink M 2003. Sex-specific consequences of recent inbreeding in an ancestrally inbred population of New Zealand takahe. Conservation Biology 17: 708–716.

Jamieson IG, Grueber CE, Waters JM, Gleeson DM 2008. Managing genetic diversity in threatened populations: a New Zealand perspective. New Zealand Journal of Ecology 32: 130–137.

Jamieson IG, Wallis GP, Briskie JV 2006. Inbreeding and endangered species management: is New Zealand out of step with the rest of the world? Conservation Biology 20: 38–47.

Jansen WP 1993. Eradication of Norway rats and rabbits from Moutohora (Whale) Island, Bay of Plenty. Ecological Management 1: 10–15.

Jones HP, Tershy BR, Zavaleta ES, Croll DA, Keitt BS, Finkelstein ME, Howald GR 2008. Severity of the effects of invasive rats on seabirds: a global review. Conservation Biology 22: 16–26.

Jordan WR, Gilpin M, Aber J 1987. Restoration ecology: a synthetic approach to ecological research. Cambridge, Cambridge University Press.

Kikkawa J 1964. Breeding density of land birds on Little Barrier Island. Physiology and Ecology 12: 127–138.

King M. 1989. Moriori: a people rediscovered. Viking, Auckland.

Krajick K 2005. Winning the war against island invaders. Science 310: 1410–1413.

Lawrence HA, Scofield RP, Crockett DE, Millar CD, Lambert DM 2008. Ancient genetic variation in one of the world's rarest seabirds. Heredity 101: 543–547.

Lovegrove TG 1996. Island releases of saddlebacks *Philesturnus carunculatus* in New Zealand. Biological Conservation 77: 151–157.

Lyver PO'B, Moller H, Thompson C 1999. Changes in sooty shearwater *Puffinus griseus* chick production and harvest precede ENSO events. Marine Ecology – Progress Series 188: 237–248.

MacAvoy ES, McGibbon LM, Sainsbury JP, Lawrence H, Wilson CA, Daugherty CH, Chambers GK 2007. Genetic variation in island populations of tuatara (*Sphenodon* spp) inferred from microsatellite markers. Conservation

Genetics 8: 305–318.
McCallum J 1980. The birds of the northern Mokohinau group. Tane 26: 69–78.
McCallum J 1982. The penetration of exotic passerines into modified forests on Little Barrier Island, northern New Zealand. Tane 28: 37–51.
MacDougall AS, Turkington R 2005. Are invasive species the drivers or passengers of change in degraded ecosystems? Ecology 86: 42–55.
McFadden I, Green T 1994. Using Brodifacoum to eradicate kiore (*Rattus exulans*) from Burgess Island and the Knights Group of the Mokohinau Islands. Science and Research Series No. 70. Wellington, Department of Conservation.
McGlone MS 2006. Becoming New Zealanders: immigration and the formation of the biota. In: Allen RB, Lee WG eds. Biological invasions in New Zealand. Berlin, Springer. Pp. 17–32.
McGlone MS, Wilson HD 1996. Holocene vegetation and climate of Stewart Island, New Zealand. New Zealand Journal of Botany 34: 369–388.
McGlone MS, Wilmshurst JM, Meurk CD 2007. Climate, fire, farming and the recent vegetation history of subantarctic Campbell Island. Earth and Environmental Science Transactions of the Royal Society of Edinburgh 98: 71–84.
McGlone MS, Wilmshurst JM, Wiser SK 2000. Lateglacial and Holocene vegetation and climate change on Auckland Island, subantarctic New Zealand. The Holocene 10: 719–728.
McKechnie S 2006. Biopedturbation by an ecosystem engineer: burrowing volumes and litter deposition by sooty shearwaters (*Puffinus griseus*). New Zealand Journal of Zoology 33: 259–256.
McLean IG, Gill BJ 1988. Breeding of an island endemic bird: the New Zealand whitehead *Mohoua albicilla*; Pachycephalinae. Emu 88: 177–182.
Merton DV, Morris RB, Atkinson IAE 1984. Lek behaviour in a parrot: the kakapo *Strigops habroptilus* of New Zealand. Ibis 126: 277–283.
Meurk CD, Foggo MN, Wilson JB 1994a. The vegetation of sub-Antarctic Campbell Island. New Zealand Journal of Ecology 18: 123–168.
Meurk CD, Foggo MN, Thomson BM, Bathurst ETJ, Crompton MB 1994b. Ion-rich precipitation and vegetation pattern on sub-Antarctic Campbell Island. Arctic and Alpine Research 26: 281–289.
Mildenhall DC 1976. Exotic pollen rain on the Chatham Islands during the Late Pleistocene. New Zealand Journal of Geology and Geophysics 19: 327–333.
Miller CJ, Craig JL, Mitchell ND 1994. Ark 2020 – a conservation vision for Rangitoto and Motutapu Islands. Journal of the Royal Society of New Zealand 24: 65–90.
Miller HC, Lambert DM 2004. Genetic drift outweighs balancing selection in shaping post-bottleneck histocompatability complex variation in New Zealand robins (Petroicidae). Molecular Ecology 13: 3709–3721.
Miller SD, MacInnes HE, Fewster RM 2009. Detecting invisible migrants: an application of genetic methods to estimate migration rates. Environmental and Ecological Statistics 3: 417–437.
Mills JA, Yarrall JW, Bradford-Grieve JM, Uddstrom MJ, Renwick JA, Merilä J 2008. The impact of climate fluctuation on food availability and reproductive performance of the planktivorous red-billed gull *Larus novaehollandiae*. Journal of Animal Ecology 77: 1129–1142.
Miskelly C 1998. Mana Island ecological restoration plan. Wellington, Department of Conservation.
Miskelly CM, Fraser JM 2006. Campbell Island snipe (*Coenocorypha* undescribed sp.) recolonise subantarctic Campbell Island following rat eradication. Notornis 53: 353–359.
Miskelly CM, Taylor GA 2004. Establishment of a colony of common diving petrels (*Pelecanoides urinatrix*) by chick transfers and acoustic attraction. Emu 104: 205–211.
Miskelly C, Timlin G, Cotter R 2004. Common diving petrels (*Pelecanoides urinatrix*) recolonise Mana Island. Notornis 51: 245–246.
Moar NT 1958. Contributions to the Quarternary history of the New Zealand flora. 1. Auckland Island peat studies. New Zealand Journal of Science 1: 449–465.
Moar NT 1959. Contributions to the Quarternary history of the New Zealand flora. 3. Pollen analysis of a peat profile from Antipodes Island. New Zealand Journal of Science 2: 35–40.
Moore JA, Bell BD, Linklater WL 2008. The debate on behavior in conservation: New Zealand integrates theory with practice. Bioscience 58: 454–459.
Moulton MP, Sanderson JG 1997. Predicting the fate of passeriform introductions on oceanic islands. Conservation Biology 11: 552–558.
Mulder CPH, Keall SN 2001. Burrowing seabirds and reptiles: impacts on seeds, seedlings and soils in an island forest in New Zealand. Oecologia 127: 350–360.
Mulder CPH, Grant-Hoffman MN, Towns DR, Bellingham PJ, Wardle DA, Durrett MS, Fukami T, Bonner KI. 2009. Direct and indirect effects of rats: does rat eradication restore ecosystem functioning of New Zealand seabird islands? Biological Invasions 11: 1671–1688.
Musgrave T 1866. Castaways on the Auckland Islands. Lockwood & Co., London.
Norton DA 2000. Benefits of possum control for indigenous vegetation. In: Montague TL ed. The brushtail possum: biology, impact and management of an introduced marsupial. Lincoln, Manaaki Whenua Pres. Pp. 232–240.
Norton DA, de Lange PJ, Garnock-Jones PJ, Given DR 1997. The role of seabirds and seals in the survival of coastal plants: lessons from New Zealand *Lepidium* (Brassicaceae). Biodiversity and Conservation 6: 765–785.
O'Dowd DJ, Green PT, Lake PS 2003. Invasional 'meltdown' on an oceanic island. Ecology Letters 6: 812–817.
Ogden J, Gilbert J 2009. Prospects for the eradication of rats from a large inhabited island: community based ecosystem studies on Great Barrier Island, New Zealand. Biological Invasions 11:1705–1717.
Ogden J, Stewart GH 1995. Community dynamics of the New Zealand conifers. In: Enright NJ, Hill RS eds. Ecology of the southern conifers. Melbourne, Melbourne University Press. Pp. 81–119.
Oliver WRB 1955. New Zealand birds. 2nd ed. Wellington, A.H. & A.W. Reed.
Parker KA 2008. Translocations: providing outcomes for wildlife, resource managers, scientists, and the human community. Restoration Ecology 16: 204–209.
Parkes JP 1990. Eradication of feral goats on islands and habitat islands. Journal of the Royal Society of New Zealand 20: 297–304.

Parkes J, Murphy E 2003. Management of introduced mammals in New Zealand. New Zealand Journal of Zoology 30: 335–359.

Prebble M, Wilmshurst JM 2009. Detecting the initial impact of humans and introduced species on island environments in Remote Oceania using palaeoecology. Biological Invasions 11:1529–1556.

Rauzon MJ 2007. Island restoration: exploring the past, anticipating the future. Marine Ornithology 35: 97–107.

Rayner MJ, Hauber ME, Clout MN 2007. Breeding habitat of the Cook's petrel (*Pterodroma cookii*) on Little Barrier Island (Hauturu): implications for the conservation of a New Zealand endemic. Emu 107: 59–68.

Rimmer A 2004. Tiritiri Matangi: a model of conservation. Auckland, Tandem Press.

Roberts CM, Duncan RP, Wilson K-J 2007. Burrowing seabirds affect forest regeneration, Rangatira Island, Chatham Islands, New Zealand. New Zealand Journal of Ecology 31: 208–222.

Robertson BC, Elliott GP, Eason DK, Clout MN, Gemmell NJ 2006. Sex allocation theory aids species conservation. Biology Letters 2: 229–231.

Russell JC, Beaven BM, MacKay JWB, Towns DR, Clout MN 2008b. Testing island biosecurity systems for invasive rats. Wildlife Research 35: 215–221.

Russell JC, Clout MN 2004. Modelling the distribution and interaction of introduced rodents on New Zealand offshore islands. Global Ecology and Biogeography 13: 497–507.

Russell JC, Clout MN, McArdle BH 2004. Island biogeography and the species richness of introduced mammals on New Zealand offshore islands. Journal of Biogeography 31: 653–664.

Russell JC, Towns DR, Anderson SH, Clout MN 2005. Intercepting the first rat ashore. Nature 437: 1107.

Russell JC, Towns DR, Clout MN 2008a. Review of rat invasion biology: implication for island biosecurity. Science for Conservation 286. Wellington, Department of Conservation.

Sandager F 1890. Observations on the Mokohinou Islands and the birds which visit them. Transactions of the New Zealand Institute 22: 286–294.

Shaffer SA, Tremblay Y, Weimerskirch H, Scott D, Thompson DR, Sagar PM, Moller H, Taylor GA, Foley DG, Block BA, Costa DP 2006. Migratory shearwaters integrate oceanic resources across the Pacific Ocean in an endless summer. Proceedings of the National Academy of Sciences of the U.S.A. 103: 12799–12802.

Simberloff D 1988. The contribution of population and community biology to conservation science. Annual Reviews of Ecology and Systematics 19: 473–511.

Simberloff D 1990a. Reconstructing the ambiguous: can island ecosystems be restored? In: Towns DR, Daugherty CH, Atkinson IAE eds. Ecological restoration of New Zealand islands. Wellington, Department of Conservation. Pp. 37–51.

Simberloff D 1990b. Community effects of biological introductions and their implications for restoration. In: Towns DR, Daugherty CH, Atkinson IAE eds. Ecological restoration of New Zealand islands. Wellington, Department of Conservation. Pp. 128–136.

Simberloff D 2002. Today Tiritiri Matangi, tomorrow the world! Are we aiming too low in invasives control? In: Veitch CR, Clout MN eds. Turning the tide: the eradication of invasive species. Gland, IUCN. Pp 4–12.

Sladden B, Falla RA 1927. Alderman Islands. A general description, with notes on the flora and fauna. New Zealand Journal of Science and Technology 9: 193–205 and 282–290.

Steadman DW 1995. Prehistoric extinctions of Pacific Island birds: biodiversity meets zooarchaeology. Science 267: 1123–1131.

Stringer IAN, Chappell R 2008. Possible rescue from extinction: transfer of a rare New Zealand tusked weta in the Mercury Group. Journal of Insect Conservation 12: 371–382.

Sykes WR 1977. Kermadec Islands flora: an annotated checklist. New Zealand Department of Scientific and Industrial Research Bulletin No. 219. Wellington, Government Printer.

Sykes WR, West CJ 1996. New records and other information on the vascular flora of the Kermadec Islands. New Zealand Journal of Botany 34: 447–462.

Taylor GA 2000. Action plan for seabird conservation in New Zealand. Wellington, Department of Conservation.

Taylor RH, Thomas BW 1993. Rats eradicated from rugged Breaksea Island (170 ha), Fiordland, New Zealand. Biological Conservation 65: 191–198.

Taylor SS, Jamieson I, Armstrong DP 2005. Successful island reintroductions of New Zealand robins and saddlebacks with small numbers of founders. Animal Conservation 8: 415–420.

Tennyson AJD 2010. The origin and history of New Zealand's terrestrial vertebrates. New Zealand Journal of Ecology 34: 6–27.

Tennyson A, Martinson P 2006. Extinct birds of New Zealand. Wellington, Te Papa Press.

Thomas BW, Taylor RH 2002. A history of ground-based rodent eradication techniques developed in New Zealand 1959–1993 In: Veitch CR, Clout MN eds. Turning the tide: the eradication of invasive species. Gland, IUCN. Pp 301–310.

Thompson DR, Bearhop S, Ross B 2005. Spread of Australasian pipit (*Anthus novaseelandiae*) onto Campbell Island following eradication of Norway rats (*Rattus norvegicus*). Notornis 52: 52–55.

Towns DR 2002. Interactions between geckos, honeydew scale insects and host plants revealed on islands in northern New Zealand, following eradication of introduced rats and rabbits. In: Veitch CR, Clout MN eds. Turning the tide: the eradication of invasive species. Gland, IUCN. Pp. 329–335.

Towns DR 2009. Eradications as reverse invasions: lessons from Pacific rat (*Rattus exulans*) removals on New Zealand islands. Biological Invasions 11:1719–1733.

Towns DR, Atkinson IAE 2004. Restoration plan for Korapuki Island (Mercury Islands), New Zealand. Wellington, Department of Conservation.

Towns DR, Broome KG 2003. From small Maria to massive Campbell: forty years of rat eradications from New Zealand islands. New Zealand Journal of Zoology 30: 377–398.

Towns DR, Ferreira SM 2001. Conservation of New Zealand lizards (Lacertilia: Scincidae) by translocation of small populations. Biological Conservation 98: 211–222.

Towns DR, Stephens T 1997. Island management and commercial sponsorship: the Mercury Islands experience. Science and Research Series No. 103. Wellington, Department of Conservation.

Towns DR, Daugherty CH, Atkinson IAE (eds.) 1990a. Ecological Restoration of New Zealand Islands.

Wellington, Department of Conservation.
Towns DR, Atkinson IAE, Daugherty CH 1990b. Ecological restoration of New Zealand islands – introduction. In: Towns DR, Daugherty CH, Atkinson IAE eds. Ecological restoration of New Zealand islands. Wellington, Department of Conservation. Pp. iii–iv.
Towns DR, Simberloff D, Atkinson IAE 1997. Restoration of New Zealand islands: redressing the effects of introduced species. Pacific Conservation Biology 3: 99–124.
Towns DR, Wardle DA, Mulder CPH, Yeates GW, Fitzgerald BM, Parrish GR, Bellingham PJ, Bonner KI 2009. Predation of seabirds by invasive rats: multiple indirect consequences for invertebrate communities. Oikos 118: 420–430.
Trewick SA, Paterson AM, Campbell HJ 2007. Hello New Zealand. Journal of Biogeography 34: 1–6.
Turbott EG 1961. Birds. In: Hamilton WM ed. Little Barrier Island (Hauturu). New Zealand Department of Scientific and Industrial Research Bulletin No. 137. Wellington, Government Printer. Pp. 136–175.
Veitch CR 2001. The eradication of feral cats (*Felis catus*) from Little Barrier Island, New Zealand. New Zealand Journal of Zoology 28: 1–12.
Veitch CR, Clout MN (eds.) 2002. Turning the tide: the eradication of invasive species. Gland, IUCN.
Walker K, Elliott G 1999. Population changes and biology of the wandering albatross *Diomedea exulans gibsoni* at the Auckland Islands. Emu 99: 239–247.
Wardle DA, Bellingham PJ, Fukami T, Mulder CPH 2007. Promotion of ecosystem carbon sequestration by invasive predators. Biology Letters 3: 479–482.
Wardle DA, Bellingham PJ, Boot KI, Mulder CPH 2009. Indirect effects of invasive predators on plant litter quality, decomposition and nutrient resorption on seabird-dominated islands. Ecology 90: 452–464.
Warham J, Wilson GJ 1982. The size of the sooty shearwater population on the Snares Islands. Notornis 29: 23–30.
Waugh S 2006 Bycatch of albatrosses and petrels in New Zealand fisheries. In: Wilson K-J ed. The state of New Zealand's birds 2006 special report: New Zealand's seabirds. Pp. 7–8.
West CJ 2002. Eradication of alien plants on Raoul Island, Kermadec Islands, New Zealand. In: Veitch CR, Clout MN eds. Turning the tide: the eradication of invasive species. Gland, IUCN. Pp. 365–373.
Wilmshurst JM, Anderson AJ, Higham TFG, Worthy TH. 2008. Dating the late prehistoric dispersal of Polynesians to New Zealand using the commensal Pacific rat. Proceedings of the National Academy of Sciences of the U.S.A. 105: 7676–7680
Wilmshurst JM, Bestic KL, Meurk CD, McGlone MS 2004. Recent spread of *Dracophyllum* scrub on subantarctic Campbell Island, New Zealand: climatic or anthropogenic origins? Journal of Biogeography 31: 401–413.
Wilmshurst JM, Higham TFG. 2004. Using rat gnawed seeds to independently date the arrival of Pacific rats and humans to New Zealand. The Holocene 14: 801–806.
Wilson HD 1987. Vegetation of Stewart Island. New Zealand Journal of Botany (Supplement): 1–80.
Worthy TH, Brassey R 2000. New Zealand pigeon (*Hemiphaga novaeseelandiae*) on Raoul Island, Kermadec Group. Notornis 47: 36–38.
Worthy TH, Holdaway RN 2002. The lost world of the moa. Christchurch, University of Canterbury Press.
Young D 2004. Our islands, our selves: a history of conservation in New Zealand. Dunedin, University of Otago Press.
Zavaleta E 2002. It's often better to eradicate, but can we eradicate better? In: Veitch CR, Clout MN eds. Turning the tide: the eradication of invasive species. Gland, IUCN. Pp. 393–403.

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
Aorangi, Poor Knights Islands	35.48 S, 174.74 E	110
Auckland	50.40 S, 166.05 E	45975
Atiu or Middle, Mercury Islands	36.64 S, 175.86 E	13
Bounty Islands	47.42 S, 179.03 E	
Breaksea	45.58 S, 166.64 E	170
Brothers Islands northern	41.11 S, 174.44 E	4
Campbell	52.53 S, 169.18 E	11216
Chalky	46.05 S, 166.52 E	475
Codfish Whenuahou	46.78 S, 167.62 E	1396
Cuvier Repanga	36.43 S, 175.77 E	170
Eastern Bay of Islands	35.21 S, 174.23 E	
Great Barrier Aotea	36.10 S, 175.41 E	27761
Great Manawa Tawhi	34.16 S, 172.14 E	408
Hauturu/Little Barrier	37.21 S, 175.89 E	3083
Hen and Chickens Islands	35.92 S, 174.73 E	
Jacquemart, Campbell group	52.62 S, 169.12 E	19
Kaikoura Selwyn	36.18 S, 175.31 E	535
Kapiti	40.85 S, 174.92 E	1970
Korapuki, Mercury Islands	36.66 S, 175.85 E	18
Limestone	35.78 S, 174.36 E	38
Long	41.12 S, 174.28 E	142
Mahuki Anvil	36.23 S, 175.29 E	45
Mana	41.09 S, 174.79 E	217
Mangere, Chatham Islands	44.27 S, 176.30 W	113
Maud	41.02 S, 174.28 E	309

Name	Location	Area ha
Mayor Tuhua	37.28 S, 176.24 E	1277
Mokohinau Islands	35.93 S, 175.09 E	
Mokoia	38.08 S, 176.29 E	135
Motuhoropapa, Noises Islands	36.69 S, 174.96 E	8
Motuihe	36.81 S, 174.94 E	195
Motuora	36.50 S, 174.80 E	86
Motutapu	35.19 S, 174.00 E	1560
Moutohora	37.85 S, 176.97 E	173
Ohinau	36.73 S, 175.88 E	43
Otata, Noises Islands	36.69 S, 174.97 E	15
Poutama	47.27 S, 167.41 E	36
Rangitoto	36.80 S, 174.86 E	2321
Rangitoto ki te Tonga D'Urville	40.82 S, 173.85 E	16782
Raoul, Kermadec Islands	29.27 S, 177.93 W	2938
Resolution	45.65 S, 166.75 E	20860
Snares Islands	48.02 S, 166.35 E	
South East Rangatira, Chatham Islands	44.35 S, 176.17 W	219
Stephens Takapourewa	40.67 S, 173.99 E	150
Stewart/Rakiura	47.05 S, 168.05 E	174600
Taukihepa/Big South Cape	47.24 S, 167.40 E	939
The Aldermen Islands	36.97 S, 176.08 E	
Tiritiri Matangi	36.61 S, 174.89 E	196
Ulva	46.93 S, 168.13 E	259
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 *Placostylus bollonsi* (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 *Hebe breviracimosa* (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 (*Streblus banksii*) (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?), *Leiopelma pakeka*
Motuara (Marlborough Sounds)	Pacific rat (g)	Okarito brown kiwi, Marlborough green gecko?, South Island robin, South Island saddleback, *Leiopelma pakeka*
Whakatere-Papanui (Marlborough Sounds)	Norway rat, Pacific rat (a)	Tuatara, Marlborough green gecko, Cook Strait giant wētā
Breaksea (Fiordland)	Norway rat (g)	Flax weevil, knobbled 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).