Turning the Tide:

The Eradication of Invasive Species

Proceedings of the International Conference

On Eradication of Island Invasives

Edited by C. R. Veitch and M. N. Clout

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The papers and abstracts published in this book are the outcome of the International Conference on Eradication of Island Invasives held at the University of Auckland, 19 to 23 February 2001. This conference sought to focus on the subject of: "Eradication of invasive species from islands; methods used and the results achieved." The term 'eradicating' included work to remove invasive species where complete eradication was some, or many, years away but the methods used were achieving positive results or providing a significant learning experience. The term 'island' included true islands, natural habitat islands (e.g. ponds), remnant and artificial habitat islands (e.g. reserves), or new invasions of natural ecosystems where eradication was deemed feasible. Preference was given to papers which provided detail of the techniques used or of the ecosystem response to the work. Significant learning experiences included methods which failed.

All papers have been peer-reviewed but, with few exceptions, the style and content is the choice of the authors. Nomenclature follows the standards accepted by the author of each paper.

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PREFACE

Turning the tide of biological invasion: the potential for eradicating invasive species

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THE THREAT TO GLOBAL BIODIVERSITY

The effects of alien invasive species on biodiversity have been described as "immense, insidious and usually irreversible" (IUCN 2000).

There is no doubt that invasive species can cause severe economic and ecological damage (Mack *et al.* 2000). They may soon surpass habitat loss as the main cause of ecological disintegration globally (Vitousek *et al*. 1997, Chapin *et al*. 2000) and are probably already the main cause of extinctions in island ecosystems. The breaching of biogeographic boundaries by the widespread, recent human transport of species has caused rapid and radical change in biological communities, including multiple extinctions. To minimise further extinctions and other ecological changes, the most important priority is to reduce the risks of new invasions. After prevention, the next priority is to eradicate existing invasive species, where this is possible. These aims are embodied in the United Nations Convention on Biological Diversity, which states that parties to this convention should " prevent the introduction of, control or eradicate, those alien species which threaten ecosystems, habitats or species".

Extinction is irreversible, but there is a growing realisation that biological invasions themselves can sometimes be reversed. With good planning, adequate techniques and sustained effort, it is now possible to eradicate many types of invasive species, especially in the early stages of an invasion, or where a population is confined to an island or limited habitat.

Turning the tide of biological invasion by eradicating invasive species can yield substantial benefits for biodiversity conservation, by raising opportunities for ecological restoration and the re-introduction of threatened species. It can also yield major economic benefits, by permanently removing the cause of damage to crops, livestock or native biodiversity, and obviating the need for costly perpetual control. Where feasible, eradication is typically more environmentally sound and ethically acceptable than long-term control. Sustained control may involve the perpetual use of toxins, trapping or shooting, and can entail more environmental risks and many more animal deaths than a brief eradication campaign. In this context, the recent successful action by animal ethicists to prevent the eradication of an incipient population of grey squirrels (*Sciurus carolinensis*) in Italy is a particularly sad example (Genovesi and Bertolino 2001). Many more squirrels will undoubtedly die in control attempts in the years to come than would have been killed in eradication of the initial population. Likely consequences of this invasion (as with so many others) are damage to crops and natural ecosystems and the decline of native species.

CONDITIONS FOR ERADICATION SUCCESS

There is a well-accepted set of conditions which must be met for the eradication of any population (Bomford and O'Brien 1995). These standard conditions for success are proper planning, a commitment to complete, putting the entire population of the target species at risk, removing them faster than they reproduce, and preventing re-invasion. Additional conditions, which are often necessary and always desirable, are support from local people and an ability to demonstrate the benefits of the eradication programme.

It is easiest to meet the necessary eradication conditions for isolated, small populations of species with low reproductive rates and no dormant life stages. Not surprisingly, the most notable successes to date have therefore involved the eradication of vertebrates (especially mammals) from isolated islands. Over the past 20 years, as techniques and confidence have improved, it has proved feasible to eradicate even quite small vertebrates from larger and larger islands. For example, in New Zealand, Norway rats (*Rattus norvegicus*) are now being eradicated from islands up to 11,000 ha in area. This is more than three orders of magnitude larger than the islands from which this species was first eradicated *c*. 40 years ago (Fig. 1).

It is fortunate that invasive mammals are among the easier species to eradicate, because they are also among the most ecologically damaging, especially on islands. Many extinctions of vulnerable birds, reptiles, and plants have been attributed to introduced mammals (Atkinson 1989), so the increasing ability to eradicate them is especially significant.

Established populations of plants, insects and other species with dormant life stages (eg. soil seedbanks) and high intrinsic rates of increase present more of a challenge for eradication, even in isolated populations. Typically, the eradication of plant populations involves a long campaign, involving the sustained removal of individuals before they set seed. In the longer term these species will be just as damaging to ecosystems as the more rapid and visible impact of many mammals.

In many situations, the feasibility of eradication will also be affected by risks to non-target species. This may prevent the use of certain techniques and limit the use of others. In some situations the risks to non-target species (including livestock, pets, crops and people) currently precludes the attempted eradication of some invasive species. However, some non-target deaths are acceptable if eradication of the invasive species is achieved and recovery of the affected non-target species is likely to be rapid. For example, in the course of the eradication of brushtail possums (*Trichosurus vulpecula*) from Kapiti Island, New Zealand, 181 birds were killed in traps, 39% of which were kereru (*Hemiphaga novaeseelandiae*) (Cowan 1992). Following the possum eradication (and subsequent eradication of rats by poisoning), the forest recovered substantially and kereru abundance rose up to six fold (Veltman 2000).

A factor that often affects the feasibility of any eradication is the dispersal abilities of the weed or pest species concerned. This affects re-invasion potential and may dictate continued vigilance even when the original population has been eradicated. For example, plants that are dispersed by birds or wind are more likely to re-invade an isolated island than those that depend on browsing mammals, gravity, explosion, or water dispersal. Human transport remains the most likely re-invasion pathway for most invasive species, emphasising the fact that prevention of invasion is of the utmost importance.

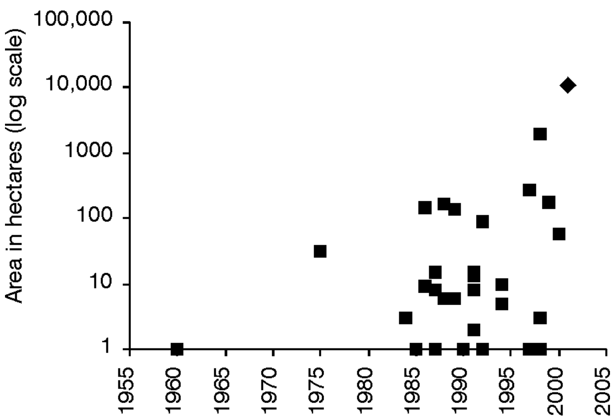

Fig. 1 Increasing size of New Zealand islands from which Norway rats (*Rattus norvegicus)* have been eradicated: 1960-2001 (square symbols) (Veitch 1995), and the yet-to-be-confirmed Campbell Island eradication (diamond symbol).

ADVANCES IN CAPABILITY AND KNOWLEDGE

There have been substantial recent advances in our ability to eradicate invasive species, exemplified by the increasing size of islands from which invasive vertebrates such as rodents have been eradicated (Fig. 1). We can anticipate more successful eradications of invasive vertebrate species, as existing technology and approaches are applied. Eradications of invertebrates and plants may pose greater challenges and require more sustained campaigns, but the general principles remain the same and there have been notable successes, some of which are described in this volume.

A topic which merits greater attention when planning eradications is that of ecosystem response to species removals (Zavelata *et al.* 2001). There may be unexpected (and sometimes unwanted) consequences of eradications, such as the ecological release of invasive plants when an introduced herbivore is removed, or irruptions of prey species after the removal of a predator. Such effects need to be borne in mind when planning eradications. Knowledge of the ecological relationships of invasive species is a key prerequisite when planning their removal from an ecosystem. These relationships raise opportunities as well as risks: for example it is possible to remove invasive prey species (e.g. rodents) and their introduced predators (e.g. cats) in a single poisoning operation, through deliberate secondary poisoning of the predators via their toxic prey.

As more eradications are attempted worldwide, it is increasingly important that lessons are learned from each and every one of these attempts (whether successful or unsuccessful) and that the information gained and skills learned are shared. This volume (and the conference on which it was based) is a contribution to the vital process of sharing knowledge to combat the threat of invasive alien species.

ACKNOWLEDGMENTS

Our aim in organising the 2001 ISSG Conference on Eradication of Invasive Species and editing its proceedings into a peer-reviewed volume was to bring together conservation practitioners and scientists who are at the forefront of the battle against alien invasive species. This volume is intended to share their insight and practical experience with a wider audience. We thank all of the participants at the

conference, especially those who have contributed papers to this volume. We also thank all those who peer-reviewed papers and assisted in other ways with its production. Special thanks are due to Carola Warner, secretary to the IUCN Invasive Species Specialist Group, who helped us throughout with the task of compiling this book

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KEYNOTE ADDRESS

Today Tiritiri Matangi, tomorrow the world! Are we aiming too low in invasives control?

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Abstract Eradication of invasive non-indigenous species is often viewed as an impossible goal and an approach historically typified by high-profile failures. However, there have been a surprising number of successful eradications of animals, plants, and even microorganisms. Although the majority of successes have concerned geographically-circumscribed invasions (e.g., on small islands), others have rid substantial continental areas of invaders (e.g., *Anopheles gambiae* from north-eastern Brazil, or smallpox from the entire Earth). Successful eradications share three features: (1) sufficient economic resources must exist for the project to be completed, (2) clear lines of authority must exist; someone must be in charge and must be able to compel cooperation, and (3) the biology of the target organism must be adequately researched and appropriate. For many but not all eradication attempts, probability of rapid re-invasion must be low for success to ensue. Further, even when the above criteria are met, an eradication attempt, even if successful, can lead to unforeseen problems, such as mesopredator release or a proliferation of non-indigenous weeds at the expense of native plants. Finally, not only can attempted eradication of widely distributed invaders be costly, but it can generate non-target impacts (e.g., on human health or species of conservation concern), the importance of which will be weighed differently by different stakeholders. Thus, successful eradication may be as much a function of political skill and public education as of technology. When eradication is feasible, a benefit-cost analysis may help indicate when it is the best management strategy. To date, eradication has been a rather idiosyncratic matter, often resting on the drive and ingenuity of one person or a few people. This has partly resulted from lack of public interest in invasions. Other developments in management of invasions should increase the appeal of eradication attempts. The evolution of more comprehensive monitoring and reporting systems, as well as more rapid response procedures, should lead to the more frequent eradication of invasions before they become metastatic. However, even invasions that escape initial elimination and spread widely may be susceptible to eradication. Many invasions that would, *a priori*, appear suitable by the above criteria for eradication have not been attacked because no one has mustered the enthusiasm to try it or generated the political support to provide the necessary resources and framework. Moreover, we do not know the geographic limits of current technologies. For example, just how great an investment would be required to rid a large island or substantial continental region of a pestiferous mammal? As with many other aspects of the invasion problem, eradication may largely be a victim of an unwarranted fatalism that could generate the very outcome that is most feared – in fact, we are not doomed to the biotic homogenisation of the Earth, but we will surely lose this war if we do not aim high.

Keywords defeatism; invasion economics; re-invasion; restoration; side effects.

INTRODUCTION

As biologists and the public worldwide increasingly recognise the damage caused by invasive non-indigenous species (Mooney 1999), they usually assume that maintenance management is the appropriate response. “Maintenance management” means controlling an invader at a density low enough that we can tolerate the damage it causes. Maintenance options typically include mechanical, chemical, and biological control, plus ecosystem management (Simberloff 2002). Although politicians occasionally call for eradication of a new invader, the total removal of every single individual remains a controversial goal (e.g., Myers *et al.* 1998), and much of the scientific community views it as a bad idea (e.g., Dahlsten 1986) for three reasons: it is seen as unlikely to succeed, it may be costly, and it may impose substantial collateral damage. Some famous failed eradications exemplify these problems. Probably the most notorious was the 14-year eradication project for the imported fire ant (*Solenopsis invicta*) in the southeastern United States (Davidson and Stone 1989), a legendary fiasco in terms of collateral damage (including to humans) and expense (over USD200 million) termed “the Vietnam of entomology” by E. O. Wilson (Brody 1975). The biology of the ant rendered successful elimination over very large areas impractical. This campaign probably worsened the fire ant invasion by causing greater mortality for its natural enemies than for the fire ant itself.

However, many invaders have been successfully eradicated (Myers *et al.* 2000; Simberloff 2001). To my knowledge, the earliest insect eradication was the elimination of the tse-tse fly (*Glossina* spp.) from the 126 km^2 island of Principe in the Gulf of Guinea (Lapeyssonie 1988). The flies were introduced in cargo from Africa in 1825, and sleeping sickness was noted beginning in 1859, ultimately reducing the human population ten fold. A four-person team completely eradicated the fly (and the disease) between 1911 and 1914. In 1956, a tse-tse fly was again noticed on Principe, and a large scientific team was immediately dispatched to the island, where they captured 66,894 flies in two months. With the aid of traps, insecticides,

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extensive brush-clearing, and massive hunting to reduce populations of pigs and wild dogs, the fly was again eradicated at a cost of £7500 and has not been seen since. Principe is an island (though not a tiny one), and many successful eradications have occurred on islands. These range from small ones, such as the elimination of the screw-worm fly (*Cochliomyia hominivorax*) from Curaçao (Baumhover *et al*. 1955), Asian citrus blackfly (*Aleurocanthus woglumi*) from Key West (Hoelmer and Grace 1989), Oriental fruit fly (*Dacus dorsalis*) from Rota and Guam (Steiner *et al*. 1955, 1965, 1970), and Pacific rats (*Rattus exulans*) from Tiritiri Matangi (Veitch 2002), to very large ones, such as nutria (*Myocaster coypus*) from Great Britain (Gosling 1989), yellow fever from Cuba (Fenner *et al*. 1988), and the melon fly (*Bactrocera cucurbitae*) from the entire Ryukyu Archipelago, including Okinawa (Iwahashi 1996; Kuba *et al*. 1996).

Though many of the most striking recent eradications have removed various mammals from islands (e.g., Veitch and Bell 1990; Chapuis and Barnaud 1995; Day and Daltry 1996a, 1996b; Pascal 1996; Day *et al*. 1998; Pascal *et al*. 1998; Varnham *et al.* 1998; Bell 1999; Donlan *et al*. 1999), successful eradication is not just an island phenomenon. The most widespread eradication eliminated smallpox from the face of the Earth (Fenner *et al*. 1988). One of the most impressive continental eradications was that of the African mosquito (*Anopheles gambiae*), a vector of malaria, from 31,000 km^2 of north-eastern Brazil (Soper and Wilson 1943; Davis and Garcia 1989). Other eradications from large parts of continents include the screw-worm (first from Florida, then from the southeastern United States, then from Mexico, and most recently from several Central American nations (Reichard *et al.* 1992; Galvin and Wyss 1996)), the cattle tick (*Boophilus annulatus*) from over a million km^2 of the United States (Klassen 1989), and bovine contagious pleuropneumonia from the United States (Fenner *et al*. 1988). For the cattle tick example, there is occasional re-invasion (see below). Eradication from smaller continental areas is fairly common, such as that of the giant African snail (*Achatina fulica*) from a region of south Florida (Mead 1979) and part of Queensland, Australia (Colman 1978), the medfly (*Ceratitis capitata*) from 20 Florida counties (references in Simberloff 1997a), yellow fever from Panama (Fenner *et al*. 1988), karoo thorn (*Acacia karoo*) from Western Australia and Victoria, and Taurian thistle (*Onopordum tauricum*) from Victoria (Weiss 1999; R. Groves, pers. comm. 2000).

Of course, besides famous failures such as the fire ant campaign, there are many other attempted eradications that have not resulted in the complete elimination of an invader; surely there are more such cases than total successes. I have not attempted a tally, because the literature is too scattered and grey, and because colloquial use of the term "eradication" makes it difficult to assess exactly what is a failure (Simberloff 1997a, 2001). Often public figures (e.g., Chiles 1996) and even scientists (e.g., Langland and Sutton 1992) use "eradication" to mean partial removal and substantial control. In these instances total eradication was never even attempted. Should such a campaign be viewed as a failure? This assessment seems unduly harsh if the same method used in the eradication campaign would have been used for maintenance management, and if substantial control results even though elimination is not complete, as in the attempt to eradicate *Spartina* spp. from New Zealand (Nicholls 1998).

In the remainder of this paper I attempt to parse the successes and failures to seek guidance as to when eradication is feasible. Do common features characterise successful campaigns? Do similar problems plague many failures? At the outset, I emphasise that I am not addressing whether society as a whole wants a particular invader removed or even controlled. Often one faction wants to eliminate a species that others see as a boon – note the battle in Australia over *Echium plantagineum*, termed Paterson's curse by ranchers and Salvation Jane by apiarists (Cullen and Delfosse 1985). Rather, assuming that society does want to control a particular species, I will ask what is the best means.

ECONOMIC RESOURCES

Eradication on a small scale may not require enormous resources; the enthusiasm and hard work of a single person or a small, non-governmental organisation may even suffice. For example, a dedicated group of scientists (the Island Conservation & Ecology Group) has succeeded in removing various combinations of feral cats (*Felis catus*), Norway and black rats (*Rattus norvegicus* and *R. rattus*), house mice (*Mus musculus*), rabbits (*Oryctolagus cuniculus*), goats (*Capra hircus*), sheep (*Ovis aries*), and burros (*Equus asinus*) from nine islands in north-west Mexico (Donlan *et al*. 1999). However, for large areas, costs are often huge. For 50 infestations of 16 plant pests of California, Rejmánek *et al.* (2000) found that log (cost) increased linearly and rapidly with log (infested area). Successful large regional eradications have been supported by significant government resources and/or private investment. The Brazilian eradication of *Anopheles gambiae* was funded by the Rockefeller Foundation and the Brazilian government (Davis and Garcia 1989), the screw-worm eradication in the United States and Mexico cost United States taxpayers USD750 million (Reichard *et al*. 1992), while the reduction of the African root parasite witchweed (*Striga asiatica*) in the Carolinas from 162,000 ha in the 1950s to *c*. 2800 ha now entailed the massive support and cooperation of the United States government and the state governments of North and South Carolina (Westbrooks 1993). Of course, huge budgets do not ensure success – witness the fire ant eradication disaster. However, for eradication over substantial areas, big budgets are generally a prerequisite (Myers *et al*. 2000; Simberloff 2001b).

The fact that expense increases rapidly as area of an invasion increases leads to the dictum that it is best to eradicate early (e.g., Simberloff 1997a; Weiss 1999; Myers *et al.* 2000). Although some longstanding, widespread invasions have been eradicated, likelihood of success is obviously improved and cost minimised if an invasion is nipped in the bud. This fact argues for effective early warning and rapid response machinery (Simberloff 1997b; Weiss 1999), a subject beyond the scope of this paper. Two cases exemplify the benefits of acting very quickly when eradication is the goal. The Caribbean black-striped mussel (*Mytilopsis sallei*), was discovered in 1999 in Cullen Bay (600 megalitres, 12.5 ha), Darwin Harbour, within six months of its arrival and before it had spread further in Australia. Within nine days the bay had been quarantined and treated with 160,000 *l* of liquid bleach and 6000 metric tonnes of $CuSO_4$. All living organisms were believed killed, and the mussel population was eradicated (Myers *et al*. 2000; Bax *et al*. 2002). The tropical alga *Caulerpa taxifolia* could almost certainly have been eliminated in the Mediterranean soon after its discovery, when it was restricted to a few square metres in front of the Oceanographic Museum of Monaco, but the effort was delayed for years and the alga now infests several thousand hectares of the coasts of Spain, France, Monaco, Italy, and Croatia (Meinesz 2001). By contrast, an effort to eradicate a small infestation of the same alga near San Diego within a year of its discovery seems promising (Meinesz 2001). An attempt to combat a much larger infestation near Los Angeles using similar methods is more problematic.

Some expenses of eradication campaigns can be substantial and not obvious at the outset (Myers *et al.* 1998). Killing the first 99% of a target population can cost less than eliminating the last 1%. This fact can become a problem with governmental funding authorities, who may be inclined to lessen support for a programme once the problem subsides, rather than see it through to completion (Schardt 1997; cf. Mack and Lonsdale 2002). Costs of monitoring may increase when pest densities are very low, yet intensive monitoring is the only effective way to determine when to end an eradication campaign. Depending on the target species and the means employed to remove it, an expensive public relations campaign may be needed to ensure public support, and lawsuits may have to be contested (Myers *et al.* 1998). For instance, for just part of a California medfly eradication project, 14,000 claims were filed for damage to car paint, and the state of California paid USD3.7 million (Getz 1989).

LINES OF AUTHORITY

It is always difficult to induce large groups of people with diverse interests to support a programme when the benefits seem unequally distributed, and eradication frequently falls in this category. Because eradication can, by its nature, be subverted by one or a few individuals, some government agency or interagency entity must have the ability to compel cooperation (Myers *et al.* 2000; Simberloff 2001b). In nations or regions with strong distrust of government, such authority will automatically generate opposition (cf. Perkins 1989). Specific concerns about the eradication techniques may be so vehement that only a strong governmental authority can enact the programme. Aerial spraying of malathion to eradicate medflies fostered widespread complaints about discomfort or threats to human health in California (Penrose 1996) and Florida (Anon. 1997). Killing large vertebrates by trapping, hunting, or poisoning often generates vocal opposition – witness the outcry over snaring feral pigs (*Sus scrofa*) in the Hawaiian islands (Van Driesche and Van Driesche 2000), trapping nutria in Great Britain (Gosling 1989), and shooting monk parakeets (*Myiopsitta monachus*) in the United States (Simberloff 1997a).

When human health is at stake, as in the tse-tse eradication on Principe or in Nigeria (Oladunmade *et al*. 1986) or the malaria mosquito eradication in Brazil, even heavy-handed government control is less likely to generate opposition. When an eradication campaign directly benefits agriculture, and the costs and possible side-effects are borne by the entire public as in spraying malathion to kill medflies, perceived inequities are more likely to generate conflict (Simberloff 2001b). Most eradications attempted for conservation purposes have occurred on small islands, often with little or no human population, and opposition has usually been minimal. Until conservation achieves a higher value in the eyes of the entire public, I predict that attempts to eradicate ecological pests over wide areas will engender hostility because of economic or emotional costs or side-effects. On a small scale, the local attempts to eradicate Asian long-horned beetles (*Anoplophora glabripennis*) by felling urban trees in Chicago and New York, and to eradicate citrus canker in Florida by destroying citrus trees, gave a foretaste of complaints that will arise if this campaign must be greatly extended (e.g., Stout 1996; Toy 1999; Sharp 2000); of course, the ultimate purpose in these instances is silvicultural or agricultural more than ecological. I know of no large-scale eradication projects conducted solely for conservation purposes, though some carried out primarily for agricultural or silvicultural reasons are perceived as having conservation benefits (e.g., that of the gypsy moth (*Lymantria dispar*)(Myers *et al*. 2000)).

BIOLOGY OF THE TARGET SPECIES

A sufficiently-determined effort can probably eradicate almost any species in a small enough area, but certain biological features can make a target less tractable. When eradication must be conducted over a large region, the biology of the target species may be particularly crucial and the scientific knowledge must be profound (Fenner *et al.* 1988; Myers *et al*. 2000; Simberloff 2001b). Some traits conducive to successful eradication are obvious – for example, large mammals are far easier to find than small insects, while plants with a soil seed bank are more difficult to eliminate than those without this feature (Simberloff 2001b). However, key biological traits often require substantial research, usually in the vein of natural history. Biological features figure large in successful eradications: smallpox has no non-human reservoir or long-term carriers (Fenner *et al.* 1988); the giant African snail does not self-fertilise (Mead 1979); *Anopheles gambiae* in Brazil was found almost exclusively near buildings (Hoelmer and

Grace 1989); while citrus canker (caused by *Xanthomonas axonopodis* pathovar *citri*), eradicated in the south-eastern United States in the early 20th century, had a very restricted host range and required movement of infected hosts by humans to spread (Merrill 1989). A recent successful eradication resting on carefully-determined biology of a pest and host was that of an introduced sabellid polychaete (*Terebrasabella heterouncinata*), parasitising abalone (*Haliotis* spp.) and other molluscs in Cayucos, California (Culver and Kuris 2000). The worms are specific to gastropod shells, especially large individuals of two common species, while the gastropod hosts have pelagic larvae, ensuring their rapid re-colonisation. The removal of 1.6 million highly susceptible hosts reduced the threshold host density below a point at which the worm could persist.

PROBABILITY OF RE-INVASION

Is the effort to eradicate an invader worth it if rapid re-invasion is likely? One reason so many eradication attempts have been on islands is that their isolation suggests immunity from rapid re-invasion. In many circumstances, even a successful eradication campaign can be a wasted effort because of re-invasion. In Washington state, an intensive campaign rid Long Lake (130 ha) of Eurasian water milfoil (*Myriophyllum spicatum*) (Thurston County Department of Water and Waste Management 1995). However, a public boat ramp permitted quick re-invasion, and the county switched to a programme of maintenance management by hand-pulling (M. Swartout, pers. comm. 1999). Other times, the probability of deliberate subversion of an eradication (Perkins 1989) is so high that the attempt may be futile. The reappearance of northern pike (*Esox lucius*) in Lake Davis, California, after its apparently successful eradication (Anon. 1999) probably resulted from sabotage (P. Moyle, pers. comm. 1999). The ease with which a single individual can subvert an eradication of some species (e.g., Davis 1990) may be an argument against the attempt when the goal is controversial.

In general, whether the probability of re-invasion should forestall an eradication campaign rests on a full assessment of the likely costs and benefits. There may be reasons to attempt eradication even if re-invasion is probable. For instance, sometimes the benefit of an eradication campaign may be a biologically artificial one, in that trade regulations may prohibit importation of some good unless its region of origin is certified as free of a pest. In such instances, the economic benefits may be so great that certain re-invasion would not argue against eradication attempts. This is the reason government officials repeatedly mount expensive eradication campaigns against the medfly in California and gypsy moth in parts of the United States and Canada in spite of a high probability of rapid re-infestation (Myers *et al*. 2000). This is not to say that the ecological and/or economic benefits of either of these campaigns might not suffice to justify them even in the absence of trade regulations. The point I am making is that low-level maintenance management, as opposed to eradication, is not an option because of trade regulations, even if maintenance management would achieve greater real control and/or cost less.

Independent of trade regulations, an eradication campaign can have sufficient economic, ecological, health, or even symbolic benefits to warrant the cost even if quick re-invasion is certain. In the successful eradication of the cattle tick from the United States, described above, re-infestation into the lower Rio Grande River region of Texas continually occurs through movement of infected animals from Mexico; leading to frequent small control operations (Klassen 1989). No one doubts the value of this programme. The Alberta rat control programme (Bourne 2000; Holubitsky 2000) is an inspirational eradication example despite frequent re-invasion. Norway rats (*Rattus norvegicus*) were first discovered on the eastern border of Alberta in 1950. Because rats destroy crops, every landowner and municipality in Alberta is mandated to destroy them, but the provincial government now pays all costs. The bulk of the activity is conducted by pest control inspectors hired and supervised by municipalities along the Alberta-Saskatchewan border. Every premise within a 29 x 600 km border zone is inspected at least annually, and control is effected primarily by eliminating food sources, extensive use of anticoagulant baits, and hunting by a team of seven provincial rat patrol officers. The cost is about CA$350,000 annually. Of course, re-invasion is continual, and every year between 36 and 216 infestations are discovered and destroyed. However, Alberta is so rat-free that discovery of a single rat in Edmonton or Calgary receives full media coverage. Aside from the benefit to agriculture of eliminating crop loss to rats, the programme has engaged the population of the entire province and sensitised them to the potential dangers of failing to deal promptly and comprehensively with invading species.

POSSIBILITY OF RESTORATION

Simply removing an invader does not constitute restoration (Towns *et al*. 1997). An ecological restoration scheme founded on eradication may be defeated by re-invasion or other problems (Simberloff 2001). Key species may be extinct and no acceptable functional equivalents available. Restoration efforts are sometimes mysteriously unsuccessful. For instance, after eradication of predators, re-introduction of stitchbirds (*Notiomystis cincta*) to New Zealand islands has failed to produce self-sustaining populations, and reasons are not apparent (Towns *et al*. 1997). Our knowledge of community structure and function is inadequate to predict with assurance the impacts of removing a prominent member of an ecological community. Thus, unforeseen impacts of eradication abound (references in Towns *et al*. 1997). Mouse densities increased greatly following eradication of Norway rats from Mokoia Island. Even control of top predator densities at levels far above eradication can lead to increases in densities of intermediate predators ("mesopredator release"; Terborgh *et al*. 1999) with various further effects throughout the community. Elimination of a predator can also lead to increased herbivore populations and damage; eradication of Pacific rats (*Rattus exulans*) from Motuopao Island to protect a native snail resulted in detrimental increases in a non-indigenous snail instead. Removal of an introduced herbivore can lead to proliferation of non-indigenous weeds rather than restoration of the native plant community. Eradication of rabbits from Motunau Island led to increases of introduced boxthorn (*Lycium ferocissimum*), while removal of grazing livestock from Santa Cruz Island (California) caused dramatic increases in fennel (*Foeniculum vulgare*) and other introduced plants (Dash and Gliessman 1994). Such changes in vegetation structure following elimination of an herbivore can, in turn, affect animal populations. For example, removal of cattle in both Nebraskan prairie (Ballinger and Watts 1995) and Mana Island, New Zealand (Newman 1994) has decreased native lizard populations by modifying vegetation.

Some impacts of eradication described in the previous paragraph might have been predicted, but others are so idiosyncratic that even a substantial scientific research project might not have suggested them. Thus, eradication is often a large, uncontrolled experiment, and we should expect unforeseen outcomes (Simberloff 2001b).

ECONOMICS OF ERADICATION

So far, I have addressed primarily the feasibility of eradicating a pest, with some attention to benefits that might accrue even if an eradication attempt is unsuccessful, as well as to unforeseen problems. I have thus avoided the key question of whether eradication is an appropriate approach even if it is feasible. Of course the prospect of permanent removal of an invader from a region, and thus the elimination of annual management costs as well as the danger of some delayed impact, must be very seductive. However, given the great costs that may be associated with successful eradication, especially over a substantial area, society cannot undertake to eradicate every pestiferous invader for which there is a high probability of eradication success. Prioritisation of invaders for management action is a general problem, and eradication decisions are just a part of that problem. Which invaders cause, or are likely to cause, the most damage, and under what circumstances is eradication the best of available management options? Typically such decisions are based on benefit-cost analyses (Arrow *et al*. 1996), but benefit-cost analyses of many natural resource issues, particularly those related to conservation, are problematic because there is often no market, as there is for an agricultural commodity (LeVeen 1989; Simberloff 1992). In the new field of invasion economics, benefit-cost analyses are especially problematic and have rarely if ever been adequately performed (Perrings *et al*. 2000). One problem is the great difficulty in predicting the trajectory of invasions, while another is the difficulty of predicting the impacts of various kinds of control measures. Surely benefit-cost analyses will have extremely wide confidence limits for many years to come.

Nevertheless, in certain circumstances, it seems that an eradication attempt would surely be justified by a comprehensive benefit-cost analysis. For smallpox (Fenner *et al*. 1988), the entire annual national and international cost of the eradication from the inception of a full-fledged campaign in 1967 to its success in 1979 was only USD23 million, while the annual cost of the disease (not counting control efforts) during this period to underdeveloped nations alone was at least USD1.07 billion, and worldwide was estimated as USD1.35 billion. The annual cost of control efforts before the eradication campaign just in the United States was USD150.2 million. Even if the campaign had not succeeded, so long as it had even a moderate probability of success, it would seem to have been an appropriate investment.

Just a rapid glance at the annual current management costs (not including losses and damages) estimated for some invaders in the United States (Pimentel *et al*. 2000) suggests that even an expensive eradication campaign might be appropriate, so long as the prospects of success were even moderate and the attempt would not substantially interfere with, or foreclose, other effective management techniques. Every year, the United States spends USD45 million on purple loosestrife (*Lythrum salicaria*) control, USD3-6 million on management of *Melaleuca quinquenervia*, USD4.6 million to manage the brown treesnake (*Boiga irregularis*) on Guam, and USD100 million to deal with Dutch elm disease (*Ophiostoma ulmi*). However, the real prospects of successful eradication of any of these species would have to be assessed based on detailed knowledge of its biology, and alternative methods (e.g., the recently released biological control agents for the first two species) may end up producing adequate control at far lower than current expenditures. My point in listing these examples is that each one entails an enormous annual expenditure, and I wonder if the possibility has been considered that total, long-lasting eradication could be achieved for, say, 10 or 20 times the current annual control cost, plus future costs of prevention. Do resource managers typically think this big?

CONCLUSIONS

There are some spectacular large-scale eradication successes. And there is a growing string of smaller successes. Further, a wide array of techniques has been successfully deployed – sterile insect release, male annihilation, traps, pathogens, vaccination, chemical sprays and baits, hunting, dogs, Judas goats, host removal, fire, and many other gory procedures. Nevertheless, eradication is almost a stepchild of management of invasives, often not considered as a possible solution even when the specifics of a situation might augur well for success. I see two main reasons for this disconnect:

- First, the literature on eradication is scattered and often very grey. Eradication of mammals is published in different outlets from insect eradication, and plant eradication histories, when published at all, are found in yet other sources. This conference is the first international conference spanning the entire field of eradication, and the number and high quality of presentations shows that the organisers have struck a very responsive chord. I predict that the published proceedings will go a long way towards both unifying the field and attracting the attention of policy makers, managers, and invasion biologists. In addition, leaders of eradication projects must recognise high-quality, international publication as a normal part of the job. If we want eradication to become a real option in managing invasive species, we have to publicise the methods and results better.

- Second, the entire problem of introduced species seems so overwhelming that it has induced a sort of fatalism – the forces arrayed against us, particularly the growing movement of cargo and people in the free-trade era, seem so overwhelming that some authors see us doomed to an eventual global homogenisation (e.g., Quammen 1998). Eradication, both because of publicised failures and because it is, in a sense, the management approach that aims the highest, falls victim to this fatalism even more acutely than other methods. But surely this sense of unavoidable doom is unwarranted. We know that eradication can work because it has. It has worked despite the relatively poor lines of communication I have outlined above and despite what would often seem to be the awesome biological powers of the target invader. New Zealanders have even developed an export industry of advice on, and application of, island mammal eradication techniques. What we do not know are the limits of most of these technologies. Just how large an island could be cleared of rodents by the techniques developed in New Zealand and northwestern Mexico? If the political will and economic support could be mustered, could nutria be completely eradicated in North America? Rabbits in Australia? What about invasive plants – under what circumstances could the witchweed approach be replicated? If smallpox and citrus canker can be eradicated, are insects on continents really out of the question?

I do not know the answers to these questions, but the inspirational stories from the literature and this conference suggest that we should not sell ourselves short. It is worthwhile to reflect on the defeatism expressed by the distinguished scientist René Dubos (1965) as he reflected on human disease eradication on the eve of the successful campaign to eliminate smallpox: "…it is easy to write laws for compulsory vaccination against smallpox, but in most parts of the world people would rather buy the vaccination certificate than take the vaccine; and they shall always find physicians willing to satisfy their request for a small fee. For this reason, and many others, eradication programs will eventually become a curiosity item on library shelves, just as have all social utopias." One thing is certain – we will surely lose the war against invasive nonindigenous species if we consider eradication an impossible fantasy and not an attainable reality.

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PAPERS

Cat eradication on Hermite Island, Montebello Islands, Western Australia

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Abstract Feral cats (*Felis catus*) and black rats (*Rattus rattus*) became established on the Montebello Islands, an archipelago of about 100 islands, islets and rocks off the Pilbara coast of Western Australia, during the late 19th century. They were probably introduced from pearling vessels. The largest island in the group is Hermite at 1020 ha. Three species of native mammals and two of birds became extinct well before the British used the islands for testing nuclear weapons in the 1950s. *Montebello Renewal* (part of the 'Western Shield' fauna recovery programme) aims to eradicate feral animals from, and reintroduce and introduce threatened animals to, the Montebellos. Rats occurred on almost every island and islet when eradication was attempted in 1996. In 1999 small numbers of rats were detected on Hermite and two adjacent islands and work is under way to eliminate them. Feral cats occurred on several islands at various times, but by 1995 were naturally restricted to Hermite. Feral cat eradication took place in 1999 and comprised two stages – aerial baiting and trapping. Aerial baiting utilised recently developed kangaroo meat sausage baits with flavour enhancers and the toxin 1080. About 1100 baits were dropped by hand from a helicopter. Hermite Island has two main soil types – sand and limestone. Aerial baiting primarily targeted sandy soils. Four cats, all females, remained after baiting. These were trapped using Victor 'softcatch'® traps set either in association with phonic and odour lures or set in narrow runways. Eradication was achieved over a six-week period. Searches for evidence of cat activity in 2000 confirmed that cats had been eradicated.

Keywords cat eradication; islands; cat bait; cat trapping.

INTRODUCTION

The importance of islands to the conservation of Australian mammal species has been well documented (Burbidge and McKenzie 1989; Abbott and Burbidge 1995; Burbidge *et al.* 1997). One of the key factors in the historic importance of islands has been that most have remained free of introduced predators. Burbidge (1999) highlighted the current and future importance of islands to nature conservation and stated that 'Australian nature conservation agencies need to pay more attention to the eradication of exotic animals from islands'.

Feral cats (*Felis catus*) pose a serious threat to populations of small to medium-sized native vertebrates. Anecdotal evidence has indicated that predation by feral cats, either acting singly or in concert with other factors, has resulted in the local extinction of a number of species on islands and mainland Australia. Burbidge and Manly (2002) analysed the relationship between disturbances and native mammal extinctions on Australian islands and implicated feral cats in the extinction of these species on arid islands. They concluded that high estimated extinction probabilities are associated with ground dwelling, herbivorous, 'critical weight range' mammals of high body weight on islands of low rainfall, low to moderate presence of rockpiles and the presence of cats, foxes and rats.

Predation by feral cats also affects the continued survival of many native species that persist at low population levels (Dickman 1996; Smith and Quin 1996) and has prevented the successful re-introduction of species to parts of their former range (Gibson *et al*. 1994; Christensen and Burrows 1995). Control of feral cats is recognised as an important conservation issue in Australia today and as a result, a national 'Threat Abatement Plan for Predation by Feral Cats' has been developed (Environment Australia 1999). The Department of Conservation and Land Management (CALM), through Project 'Western Shield', has been working over the past few years to develop an effective cat control strategy. *Montebello Renewal* (part of 'Western Shield'), which aims to eradicate rats and cats and to reintroduce locally extinct species, provided an opportunity to assess the effectiveness of these techniques to eradicate cats from an island.

The Montebello Islands comprise a group of over 100 islands, islets and rocks off the Pilbara coast of Western Australia. The archipelago has a tropical, arid climate. The nearest weather station is on Barrow Island, 30 km to the south, which has a median rainfall of 285 mm, and mean daily maximum and minimum temperatures of 30.3°C and 21.4°C respectively.

Montague (1914) conducted the first detailed biological survey of the islands in 1912. He observed the presence of cats and noted that they had probably established from a shipwreck 20 or so years before his visit. It seems more likely, however, that cats were introduced from pearling vessels that were active in the area from the 1860s. Montague attributed the recent extinction of the golden bandicoot (*Isoodon auratus*) to predation by cats and predicted that the spectacled hare-wallaby (*Lagorchestes conspicillatus*) would suffer the same fate. Later surveys by Sheard (1950) and Serventy and Marshall (1964) found

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that both species had become locally extinct on the islands, confirming Montague's prediction.

The above surveys recorded cats on Hermite Island, at 1020 ha the largest island in the group. However, cats were also observed on Trimouille Island in 1970 (Burbidge 1971) and tracks were recorded by K. D. Morris on Bluebell Island in 1985 (Burbidge *et al.* 2000). Surveys between 1994 and 1996 found that cats were then restricted to Hermite Island, indicating that populations on the smaller islands had died out without human intervention (Burbidge *et al.* 2000).

Montebello Renewal aims to eradicate feral cats and black rats (*Rattus rattus*) from the Montebello Islands to allow the successful re-introduction of native mammal species and also two species of locally extinct birds: spinifexbird (*Eremiornis carteri*) and the black-and-white fairy-wren (*Malurus leucopterus leucopterus*) (Burbidge 1997). The absence of cats and eradication of rats from Trimouille Island has allowed this island to be used for the introduction of species threatened with extinction on mainland Australia. The mala (*Lagorchestes hirsutus* unnamed central Australian subspecies), which is 'extinct in the wild' and is subject to predation by feral cats, has been successfully established on Trimouille Island (Burbidge *et al.* 1999, 2000; Langford and Burbidge 2001). The djoongari (Shark Bay mouse, *Pseudomys fieldi*), also threatened by feral cat predation, was introduced to North West Island in June 1999 and August and October 2000.

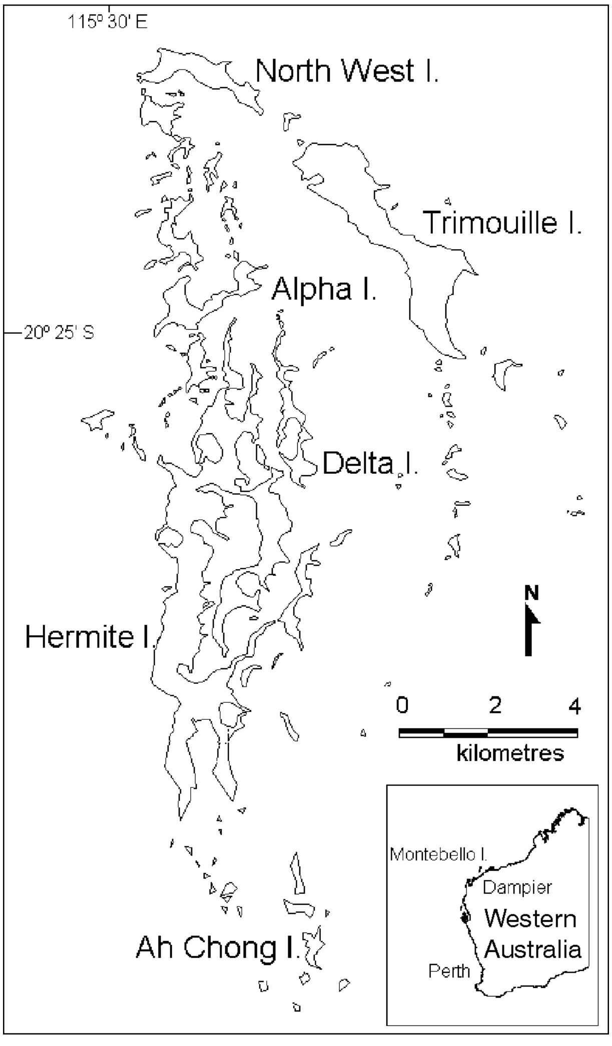

Fig. 1 Montebello Islands, showing the location of Hermite Island.

METHODS

Site Description

The Montebello Islands are located between 20º21' and 20º32' South and between 115º31' and 115º36' East, approximately 100 km off the Western Australian coast. The total area of the islands is approximately 2300 ha with Hermite Island being the largest at 1020 ha. Hermite Island is a difficult location on which to conduct a cat eradication campaign because of its isolation, rugged terrain and absence of vehicle access. The shape of the island is elongated and highly convoluted, with a number of sandy beaches, areas of mangroves, cliffs and limestone ridges and peninsulas (Fig. 1). Its interior is low, undulating and is vegetated with a dense mat of spinifex (*Triodia* sp.) with occasional *Acacia coriacea* thickets on deep sand. Access was via small boat along Stephenson Channel and then on foot, carrying the traps and trapping equipment.

Cat Eradication Strategy

The programme to eradicate feral cats on Hermite Island involved aerial baiting to remove the majority of the cats, followed by intensive trapping, if warranted, to remove the remaining individuals. A reconnaissance of Hermite Island was conducted prior to the baiting programme to assess cat abundance. Searches for evidence of fresh cat activity were conducted around most mangrove stands and sandy areas on the island. These were examined daily over a five day period. The location of fresh cat activity on swept areas, its extent and the distances between sites suggested that at least 20 cats were present prior to baiting.

Baiting Programme

CALM researchers have recently completed development of a bait to control feral cats. The bait is similar to a chipolata sausage. It is 20 g in weight and contains a number of flavour enhancers that are highly attractive to feral cats (Patent No. AU 13682/01). The baits were manufactured at the Department's Bait Factory and airfreighted to the island. At the island the baits (National Registration Authority experimental baiting permit No. 1213) were prepared for laying by thawing and then blanching (that is, placing in boiling water for one minute). The toxin 1080 (sodium monofluoroacetate) was injected into the baits at a rate of 3.0 mg/bait. A risk analysis concluded that there are unlikely to be any significant effects on non-target species on the island. All baits were treated with an ant deterrent compound (Coopex®) at a concentration of 12.5 g /*l* Coopex as per the manufacturer's instructions. Ant attack on baits rapidly degrades the bait medium, reducing palatability, and the persistence of ants on the bait deters uptake by feral cats (D. Algar, pers. obs.).

About 1100 baits were dropped by hand from a helicopter on the 3 July 1999. The flight path followed the 140 km coastline and then through the centre of the island to maximise bait availability and the area covered.

Trapping Programme

It is unlikely that a single baiting campaign will achieve eradication of cats within an area of this size. Monitoring cat activity along a number of the beaches post-baiting indicated that several cats were still present. To remove the remaining cats a trapping programme was started ten days after the baiting campaign.

Trapping systems for cats have generally relied on food-based lures as the trap attractant (reviewed by Veitch 1985). A number of other olfactory scents or social odours to entice cats into traps or bait stations have also been used (Veitch 1985; Clapperton *et al.* 1994; Edwards *et al*. 1997). An alternative technique to these systems, using lures that mimic signals employed in communication between cats, has been developed by CALM researchers and proven highly successful. Cats are very inquisitive about other cats in their area; their communication traits are principally reliant on audio and olfactory stimuli. The trapping technique utilises padded leg-hold traps, Victor 'Soft Catch'® traps No. 3 (Woodstream Corp., Lititz, Pa.; U.S.A.), a Felid Attracting Phonic (FAP) that produces a sound of a cat call, and a blended mixture of faeces and urine (Pongo). Each trap site consists of a channel of approximately 40 cm wide and 80 cm in length, cleared into a bush to create a one-way (blind) trap set. Two traps, slightly offset (approximately 2-5 cm), are positioned at the entrance of the blind set, at each trap site. The free jaws of the two traps are aligned in the centre and almost touching. A trap bed is made so that when lightly covered with soil, the traps are level with the surrounding ground surface. A guide stick is placed in front of the traps to force animals to lift their foot then push down onto the pressure plate. Both traps are secured in position by a 30 cm length of chain to a 30 cm steel anchor peg. A 12 x 8 x 2 cm foam pad is placed below the pressure plate to prevent soil from falling into the trap bed and compacting under the plate. The traps are then lightly covered with soil.

Cats are lured to the trap set initially by the audio signal produced by the FAP. The FAP is located at the back of the trap set, either concealed under leaf litter or hidden within the bush. The FAP consists of a 36 x 25 mm printed circuit board with a microprocessor data driven voice ROM. As cats approach the trap set they are further enticed into the traps by the smell of 'pongo'. The pongo consists of a blended mixture of cat faeces and urine in a ratio of approximately 1:1. Approximately 20 ml of this mixture is placed in a shallow depression about 30 cm from the centre of the trap plates.

Trapping campaigns can sometimes induce trap-shyness in the target species; trapping for feral cats is no exception. Variations on the standard trap set were used towards the end of the trapping operation to capture remaining cats wary of the standard set. The most successful variation of the trap set was a 'road trap' that involved placing four traps in a set along pathways actively used by cats.

Five personnel (two professional trappers, two volunteer trapper assistants and a boat handler) were involved in the trapping programme after the first week. Personnel were rotated every two weeks; however, three of the trappers stayed longer. The entire trapping exercise took six weeks to complete.

The difficult terrain and distances to be walked every day precluded trapping the entire island simultaneously. The island was therefore divided into four zones: east, north, west and south. Each of these zones terminated in a sandy narrow-necked peninsula that could be used to assess cat movement into the area once trapping had been terminated. The trapping programme was initially conducted on east and south Hermite and as traps were removed, trapping commenced on west and north Hermite. Traps are normally placed at 1 km intervals along tracks; however on Hermite more effort was put into providing broad-scale trap coverage and maximising trap success. Traps were located strategically on all areas of perched sand sheet and dunes across each zone. Additional 'road traps' were located in areas where cats had not entered the standard set. In total, 180 trap sets, totalling 1544 trap-nights, were placed over the island during the trapping period.

The traps were left in position for a minimum of seven days and if no cat activity had been recorded in the zone, the traps were removed. Evidence of fresh cat activity was recorded for each trap site and intensive searches were conducted *en route*. Once trapping in each zone was completed, the area was searched carefully to ensure that all cats had been removed. The sandy areas that terminated each zone were monitored on a regular basis to ascertain whether cats had moved into previously trapped zones.

Trapped cats were humanely destroyed; then sexed and weighed. Stomach contents were collected for diet analysis and the females were examined for placental scarring.

RESULTS

The intensive searches showed that cats had been active across much of the island, mostly along the sandy beaches, mangroves and *Acacia* thickets where 'highways' of tracks and numerous scat piles were observed. Some evidence of cat activity was observed along the limestone ridges and in the spinifex plains, but these areas were understandably less favoured habitat and were used as pathways to the more preferred sites.

Four cats were captured during the trapping programme. Two cats were trapped on the standard audio and scent lure and the remaining two in 'road traps'. All four cats entered the standard trap set on first encounter, although on two occasions the traps did not trigger. These two animals did not enter standard trap sets again and their capture required the placement of road traps. Since this trapping programme, improvements in trap maintenance and modification of the trap set have resulted in capture of all cats entering the standard audio and scent trap set. The modification to the trap set involves making the channel slightly wider than the width of one trap and then positioning the two traps one in front of the other at the entrance of the blind set.

No evidence of fresh cat activity was observed across the island once the four cats had been trapped and it was concluded that eradication had been successfully completed. This was confirmed by searches for cat activity in August 2000. The fact that only four cats remained after baiting indicates that it was responsible for removing at least 80% of the cats from the island.

DISCUSSION

Cat eradication programmes on islands are usually conducted using a combination of baiting, trapping and hunting (Veitch 1985; Rauzon 1985; Bloomer and Bester 1992; Bester *et al.* 2000). These eradication programmes have met with varied success, their success and time to completion having been limited in part by lack of effective bait and trap lures.

Bait acceptance by feral cats is in part related to the abundance of prey species (D. Algar and J. Angus pers. obs.). The major prey items available to cats on the island would have been rats, birds, reptiles and insects. The baiting campaign on Hermite Island was conducted when rat numbers were very low after an unsuccessful rat eradication project and when the availability of natural prey items, particularly reptiles and insects, was likely to be at its lowest (mid- to late-winter). Further research being conducted on the timing and frequency of baiting programmes should improve their effectiveness and cost efficiency. The cat eradication programme on Hermite Island was achieved in a matter of weeks and could have been completed sooner with the subsequent modification to the trap set. Elsewhere in the world, cat eradication projects on islands have often taken months or years, or are still ongoing. However, it is difficult to compare the efficacy of our baiting and trapping programme with others, which have taken place on islands of different climate, terrain, shape and size and with different prey availability. Some islands, for example, Macquarie Island of 11,800 ha, where eradication has not been achieved, are much larger than Hermite Island.

Feral cat eradication programmes for a number of islands off the Western Australian coast are now being planned. Targets for the future include Faure Island (5200 ha – reconstruction of original fauna plus marooning), Garden Island (1050 ha – protection of native animals including tammar wallabies (*Macropus eugenii*) and carpet pythons (*Morelia spilotus*)) and Dirk Hartog Island (58,600 ha – reconstruction of the original fauna). The advances in cat control strategies developed by the Department may also be useful in assisting eradication of feral cats from many islands around the world. Eradication of feral cats has already commenced on the Cocos (Keeling) Islands (1400 ha) in the Indian Ocean.

Black rats are still present on Hermite Island (although eradicated from all other islands in the group). Once they have been eradicated the reconstruction of the original fauna of Hermite can commence.

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Eradication of introduced *Bactrocera* species (Diptera: Tephritidae) in Nauru using male annihilation and protein bait application techniques

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Abstract Four introduced *Bactrocera* species were recorded in the Republic of Nauru in 1992. A programme to eradicate the four species was implemented between October 1998 and December 2000. The objectives were to eradicate the introduced pest fruit flies that were a threat to neighbouring Pacific Island countries and territories, to test the efficacy of Fipronil as an alternative toxicant to malathion for the management of fruit flies, to train national plant protection and quarantine staff in fruit fly eradication and emergency response techniques, to establish and up-grade the quarantine services in Nauru, and to increase fruit availability for local consumption. A combination of male annihilation and protein bait application techniques was used for eradication. The Male Annihilation Technique involved distributing fibreboard ('Canite') blocks impregnated with male fruit fly lure (methyl eugenol and/or cue-lure) and the insecticide Fipronil in a loose grid, resulting in at least 300 blocks per km² over Nauru. The blocking campaigns were repeated every eight weeks from late October 1998. The protein bait application technique involved spraying host fruit trees in hot spot areas with protein insect lure and Fipronil gel on a weekly schedule. Three of the four species, namely oriental fruit fly (*Bactrocera dorsalis*), Pacific fruit fly (*B. xanthodes*), and melon fly (*B. cucurbitae*), were declared eradicated. Populations of mango flies (*B. frauenfeldi*) still persist. The Government drafted and promulgated a new Agricultural Quarantine Act and established an Agricultural Quarantine Service in Nauru. A major benefit of the eradication programme is that people in Nauru once again are able to eat mangoes and breadfruit after a decade of near-total losses due to introduced fruit flies.

Keywords Tephritidae; fruit flies; eradication; male annihilation technique.

INTRODUCTION

Fruit flies (family Tephritidae) pose a significant threat to fruit and vegetable production and to the unimpeded export of fresh fruits and fleshy vegetables throughout the world. In the Pacific region, over the past 25 years, exotic fruit flies invaded several countries causing direct losses to production of fresh fruits, imposition of trade restrictions by importing countries, and implementation of expensive eradication or suppression programmes to rid countries or parts of countries of the introduced pests. For example, Asian papaya fruit fly (*Bactrocera papayae* (Drew and Hancock)) gained entry to Papua New Guinea (PNG) in about 1992 and was recorded in northern Australia near Cairns in 1995 (Drew 1997). The outbreak in the Cairns area was subsequently eradicated at a cost of about AU$35 million. Other members of the *dorsalis* complex of fruit flies gained entry into several areas of the Pacific region. Oriental fruit fly (*B. dorsalis* (Hendel)) expanded its geographical range into Tahiti and Moorea in French Polynesia in about 1996. In 1997, *B. philippinensis* Drew and Hancock was recorded in, and subsequently eradicated from, the Darwin area of the Northern Territory of Australia. *B. occipitalis* (Bezzi) and *B. philippinensis* were recorded in the Republic of Palau in Micronesia in 1996. Allwood *et al*. (1999) and McGregor (2000) examined the technical and economic feasibility of eradicating these species and, subject to funding support, an eradication programme may commence in October 2001. Melon fly (*B. cucurbitae* (Coquillett)) was introduced into the Western Province of Solomon Islands around 1984 and now has spread as far south as Guadalcanal in the Central Province of Solomon Islands (Hollingsworth *et al*. 1997). Mediterranean fruit fly (*Ceratitis capitata* (Wiedemann)) was recorded in New Zealand in 1996 and was successfully eradicated at a cost of approximately NZ$6 million.

Staff of the South Pacific Commission (now called the Secretariat of the Pacific Community (SPC)) conducted a survey of fruit flies in the Republic of Nauru in November 1992, using five pairs of modified Steiner traps located in urban, village, secondary forest, and beach areas and on a vegetable farm. One trap of each pair was baited with methyl eugenol plus 50% malathion emulsifiable concentrate in a ratio of 3:1 by volume and the other baited with cue-lure and malathion. This survey recorded that four species of fruit flies were established. The introduced species were oriental fruit fly, Pacific fruit fly (*B. xanthodes* (Broun)), melon fly, and mango fly (*B. frauenfeldi* (Schiner)), the first two species being attracted to the male lure methyl eugenol and the last two being attracted to cue-lure (H. Kumar pers. comm.). Chu (1993) of the National University Taiwan trapped large numbers of oriental fruit flies and melon flies in the east, southeast and Buada Lagoon areas of Nauru.

Mango fly is widespread in Micronesia (except in Guam and Commonwealth of the Northern Mariana Islands (CNMI)), PNG, Solomon Islands, and northern Queens-

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land. Pacific fruit fly is widespread from Fiji Islands east to Cook Islands. In contrast, the regional distribution of melon fly is restricted to PNG, Guam, CNMI, Solomon Islands and Nauru and the oriental fruit fly to French Polynesia and Nauru. Both mango fly and oriental fruit fly have very wide host ranges and, as adequate fruits were available at most times of the year, high populations were present at all times in Nauru. Pacific fruit fly was restricted primarily to *Artocarpus* spp. and, consequently, high populations of the fly occurred in October to April. Melon fly is restricted mainly to hosts belonging to the family Cucurbitaceae. Melon fly populations were generally low, but present throughout the year.

Regional organisations, such as the SPC, Food and Agriculture Organisation of the United Nations (FAO), and the United Nations Development Programme (UNDP), and the Governments of the Pacific Island countries and territories (PICTs) strongly recommended that oriental fruit fly and melon fly be eradicated from Nauru, for several reasons. Eradication would reduce the threat that these damaging fruit fly species posed to fruit production and export in neighbouring PICTs, protect advances made in regional management of fruit flies since 1990, and provide an ideal opportunity to facilitate hands-on training in fruit fly eradication techniques and emergency response planning to many plant protection and quarantine staff in the Pacific region. Also, eradication of oriental fruit fly, in particular, would increase the availability of fresh fruits in Nauru, a very scarce resource since at least 95% of fruits, such as mangoes and guavas, were infested with fruit fly maggots and inedible. To protect the investment of the eradication effort, the Government of the Republic of Nauru was strongly encouraged to draft and promulgate its first Agricultural Quarantine Act and develop a small, appropriately trained Agricultural Quarantine Service to ensure fruits entering Nauru were free from damaging fruit flies and other exotic pests.

The public of Nauru reacted adversely to the prospect of using malathion as the toxicant for eradication, primarily because of its unacceptable odour. For this reason, Fipronil [(±)-5-amino-1-(2,6-dichloro-K, K, K-trifluoro-p-tolyl)-4-trifluoromethylsulfinyl-pyrazole-3-carbonitrile], a product of Aventis CropScience Pty Ltd, was selected as an alternative toxicant and was laboratory and field-tested for use in the eradication programme.

Nauru proved to be an ideal place to conduct an eradication programme for fruit flies. It is an isolated island, so the risk of re-introduction of exotic fruit flies was low. It is 41 km south of the Equator at 166º 56´ East longitude. Nauru is about 650 km south-west of Kosrae in Federated States of Micronesia and a similar distance almost due west from Tarawa in Kiribati, both of which are infested with mango fly. It is about 1000 km northeast from Honiara in Solomon Islands, the closest infestation of melon fly. The closest infestation of oriental fruit fly is in Hawaii, about 3600 km northeast of Nauru.

Nauru is 5.5 km from north to south and 4.5 km east to west and covers an area of 21.2 km², with a coastline of 19.3 km. It is an uplifted limestone island with a narrow coastal belt encircling a limestone escarpment reaching 30-70 m above sea level. Much of the escarpment and the interior of the island (referred to locally as Topside) are inaccessible due to severe land disturbances caused by extensive phosphate mining. The Buada Lagoon area in the central southwest of the island is fertile, surrounds a small brackish lake, and supports small groves of mango (*Mangifera indica*), guava (*Psidium guajava*), and breadfruit (*Artocarpus altilis*) trees. Soursop (*Annona muricata*) forms an understorey in most of the Buada Lagoon area. The Buada Lagoon area is a residential area with fruit trees growing in backyards. Nauru is located in the dry belt of the equatorial oceanic zone, with a mean daily temperature range of 26-32°C and an average annual rainfall of 1500 mm (range of 300-4572 mm). Long droughts are common, often causing the death of native trees, wild cucurbits, and breadfruit trees. The flora is poor relative to other Pacific islands, partly due to the mining activity. The range of host fruits for fruit flies is limited to plants such as Pacific almond (*Terminalia catappa*), *Guettarda speciosa*, wild guavas (*Psidium* spp.), mango, soursop, breadfruit, *Citrus* spp., and mountain apple (*Syzygium malaccense*).

This paper summarises the methods used in eradicating three species of fruit flies in Nauru, the results obtained, and the technical and management lessons learnt during the eradication operations.

METHODS

The techniques available for the eradication of fruit flies worldwide usually follow an integrated approach, including fruit movement controls, destruction of fallen and unwanted fruits, biological control using inundative releases of parasitoids, protein bait application, application of systemic larvicides to fruit trees to kill eggs and larvae of fruit flies, ground application of an insecticide to kill larvae and emerging adults, male annihilation, and release of sterile flies. In the case of the eradication programme in Nauru, the major techniques selected were managing fallen fruits, developing adequate quarantine regulations to prevent re-introduction of exotic fruit flies, male annihilation technique (MAT), and protein bait application technique (BAT). Other techniques were unacceptable environmentally (e.g., ground application of insecticide or cover spraying trees with systemic insecticides) or were economically or technically inappropriate for a small island such as Nauru (e.g., sterile insect technique).

Organisation

The Nauru Fruit Fly Eradication (FFERAD) Programme belonged completely to the Government of the Republic of Nauru, with technical and financial support being provided through the FAO/UNDP/AusAID/SPC Project on Regional Management of Fruit Flies in the Pacific

(RMFFP). Financial support was also received from the Crawford Fund for International Agricultural Research. Aventis CropScience provided Fipronil products at no cost to the programme. The Nauru Government arranged teams for blocking from the Departments of Youth, Health, and Works, and from the Nauru Phosphate Corporation and the Buada Lagoon community. The Department of Island Development and Industry provided the coordination role and staff for the treatment of blocks, supervising MAT and BAT operations, servicing of traps, fruit sampling and processing, public awareness, and reporting on progress. Staff from the RMFFP assisted with the monitoring of the operations and progress of the eradication programme, with a special focus on quality assurance for the treatment of blocks and the distribution of blocks and protein bait.

Effective public awareness and cooperation were recognised as being essential to the success of the programme and were carried out by producing a FFERAD Newsletter during each blocking campaign and distributing these to all government departments, the Nauru Phosphate Corporation, the general public, and to school children. Public meetings and regular briefings of government personnel and the public were undertaken.

Management of fallen or unwanted fruits

Destruction of fallen, over-ripe, or unwanted fruits was not practised in the true sense, although destruction of fruits was strongly encouraged through publicity programmes. However, community leaders actively encouraged children and adults not to climb mango trees and other fruit trees to shake branches to collect fruits. When the practice of shaking branches to collect fruits was stopped, there were fewer fruits left on the ground as egg-laying sites for oriental fruit flies and mango flies. The public was encouraged by community leaders to take only fruit that had fallen to the ground or that was obviously ripe on the trees and to harvest sufficient fruits for their use rather than discarding unused fruits. The public was encouraged not to plant cucurbits until after the melon fly was declared eradicated.

Despite this effort, wild fruits such as Pacific almond and *G. speciosa*, were not collected and destroyed and, as a result, significant breeding sites for mango fly, in particular, were available. This occurred especially on Topside, where individual plants or small clusters of both host species were present and virtually inaccessible to people carrying out treatments.

Male annihilation technique (MAT)

The MAT aims to reduce the male fruit fly population to such a low level that no mating occurs. This may be achieved by distributing, at regular intervals over a wide area, a carrier containing a male lure plus a toxicant. The effectiveness of the MAT may be severely reduced if the carrier loses its attractiveness or toxicity before the end of the interval selected. Carriers may be made of fibreboard blocks (Steiner and Lee 1955), coconut husk blocks (C. Garnier pers. comm.), paper mâché discs or rectangles (R. Bull unpublished), pieces of cotton string or rope (Bateman 1982), or thickened gels (Cunningham *et al*. 1975). In Nauru, fibreboard blocks (50 mm x 50 mm x 12.7 mm) were chosen as the carrier. The blocks were cut from sheets of Standard Canite (supplied by Pacific Islands International, Kirwan, Queensland). Each sheet (2440 mm x 1220 mm x 12.7 mm) produced about 1000 blocks. Approximately 10% of the sheet was lost during the cutting process done by using a circular saw at the Nauru Phosphate Corperation workshops.

The formulation of Fipronil used was a special proprietary premix, provided by Aventis CropScience Pty Ltd in Brisbane. It contained 278 g active constituent/*l* initially, but this was thought to be slightly unstable and may have resulted in crystals settling out. The final premix contained 250 g active constituent/*l*. Initially, it was used at the rate of 3.1 ml/*l* of male lure, but this rate was changed to 4.0 ml/*l*. This premix is virtually insoluble in water, so would be very stable in the block. Fipronil is virtually odourless and so this feature overcame one of the major concerns of the public in Nauru. Laboratory bioassays conducted in Fiji Islands on Pacific fruit fly and in Brisbane on Queensland fruit fly (*B. tryoni* (Froggatt)) showed that Fipronil was effective in killing these fruit fly species at low dosages (R. Bull unpublished). Also, observational evidence indicated that, because Fipronil takes several hours to kill flies, an added advantage of transmission of Fipronil from males to females during mating might exist.

The treatment of fibreboard blocks with the male lure and Fipronil was done in used 200 *l* steel drums cut longitudinally to form 100 *l* troughs. The blocks were loosely packed into rectangular baskets covered in galvanised chicken mesh, which fitted into the 100 *l* troughs. Approximately 800 blocks were treated at one time. A mixture of male attractant and Fipronil was poured over the blocks in the trough, with the excess that drained into the trough being ladled over the blocks until the blocks had absorbed a prescribed amount of the mixture (see *Stages 1* and *2* on page 23). Random samples of 100 blocks were taken at intervals during the treatment and weighed to determine when sufficient lure plus Fipronil had been absorbed per block. Blocks were also examined for absorption by breaking them in half. The blocks were allowed to drain in the 100 *l* troughs before being stored in plastic garbage bins. Blocks were nailed with 50 mm steel nails. Galvanised nails were not used because reports from the eradication programme for Asian papaya fruit fly in northern Queensland indicated that phytotoxicity to some palm trees occurred (R. Drew pers. comm.).

Nauru was subdivided into seven sectors, five around the coastline covering residential areas, the Buada Lagoon area, and the mined area or Topside. The areas of these sectors were 0.7-1.7 km², with the exception of Topside, which was 14 km². Each sector was allocated to a team to distribute the blocks treated with male lures and Fipronil. Initially, there were sufficient teams for each team to be responsible for a particular sector, but, as the number of teams decreased to four or five, teams had to cover more than one sector. The aim of the MAT was to cover all sectors and distribute the blocks in about 1.5-2 days every eight weeks. Each team was responsible for mapping, on a daily basis, the area covered by blocking, the number of blocks distributed, and any complaints from the public.

The aim of the programme was to distribute the blocks at a minimum density of 300-400 blocks per km² over the whole of Nauru. In areas that were readily accessible by ground teams, the objective was to nail one block to a tree in the shade of foliage at a height of at least 2 m on a grid of about 50 m. If this were achieved, the density of blocks would be about 400 blocks per km². However, in urban or village areas and in areas of high incidence of fruit flies (e.g., in the Buada Lagoon area), blocks were distributed at densities much higher than required (i.e., up to 1500-1700 blocks per km²). Generally, however, the density of blocks in urban and accessible native vegetation areas was acceptable at 400-700 per km². In the accessible mined area of Topside, blocks were distributed at 50 m intervals along all roads, train tracks, and motorcycle and walking tracks that radiated from a central point and along roads that ran around the coastal edge of the escarpment. In mined, inaccessible areas on Topside, blocks were thrown or fired from slingshots into native vegetation patches. The density of blocks on Topside was, because of the terrain, 60-135 per km².

Protein bait application technique (BAT)

The principle of BAT uses the nutritional need of female fruit flies for protein before they are capable of laying viable eggs. Sexually immature female flies actively seek protein sources such as bacteria and other exudates on the leaf and fruit surfaces of the host trees. Male and sexually mature female flies also feed on protein. Adding a toxicant to the protein and applying the mixture in large droplets or spots to the underside of leaves in host trees is a very effective method of controlling fruit flies by killing female flies before they reach the egg-laying stage. Used on its own, or preferably as an adjunct to MAT, BAT is an effective eradication technique. Aventis CropScience Pty Ltd in Brisbane developed the protein bait gel used in Nauru. It comprised a mixture of protein autolysate, called Mauri Pinnacle Protein Insect Lure – Low Salt (420 g of protein/*l*) (MPPIL) (supplied by Mauri Yeast Australia, Toowoomba, Australia), Fipronil gel powder, and water. The Fipronil gel was prepared by sprinkling the Fipronil gel powder on the surface of water at the rate of 5 g of powder per litre of water and stirring vigorously until a thickened gel was formed. Protein autolysate was mixed with the gel at the rate of 30-50 ml of MPPIL per litre of gel immediately before use.

This mixture was applied to the undersides of foliage of host trees in spots of 10-15 ml at a rate of 25 spots per hectare. For each treatment, 180-200 *l* of protein/Fipronil gel bait were applied per week, providing coverage of 480-800 ha. Application was supposed to be done weekly, especially in areas where persistent fruit fly populations (commonly referred to as 'hot spots') occurred. However, due to problems of staff availability and commitment and non-arrival of supplies owing to inadequate planning and lack of regular air and sea freight services, treatments were not as consistent as they should have been. Although applications were done in November 1998 and June 1999, the main body of applications did not start until July 1999 and, even then, there were interruptions to the spray schedules in September and December 1999 and at various times during 2000.

Several types of pressurised sprayers were tested during the programme, but the most suitable and durable was the single-action 'Rega' sprayer made of brass, with a 5 *l* chemically resistant plastic container, with a sling for carrying on the shoulder.

Fruit fly monitoring procedures

Trapping

The number of modified Steiner traps (Drew 1982) for monitoring fruit fly populations varied as the programme progressed and as the numbers of flies decreased. The numbers of traps increased from 10 in October 1998, to 12 in February-May 1999, to 25 in May-September 1999, and to 41 in October 1999. This final trapping density represented two traps per km². Each site consisted of a pair of traps. One trap in a pair was baited with methyl eugenol plus malathion (50% emulsifiable concentrate) mixed in a ratio of 3:1 and the other with cue-lure and malathion in the same ratio. The traps were cleared weekly and the flies were identified and counted. Lures and insecticide were replenished every eight weeks, coinciding with the commencement of each blocking operation. No liquid protein traps (Drew 1982) were used to assess the numbers of female flies; this feature was possibly a deficiency in the programme.

Fruit sampling

Sampling of the major fruits were undertaken periodically to assess the percentage of fruits infested by the respective species. The sampled fruits covered 15 plant families and 19 species, including mango, soursop, papaya (*Carica papaya*), guava, mountain apple, lime (*Citrus aurantifolia*), vi (*Spondias dulcis*), Pacific almond, *G. speciosa*, *Ochrosia elliptica*, bitter gourd (*Momordica charantia*), *Calophyllum inophyllum*, *Hernandia* sp., *Ficus* sp., and *Morinda citrifolia*. Large samples of fruits were randomly collected, mainly from the ground, weighed, counted, and set up in bulk or individually in separate plastic containers over sieved sawdust. They were held in an air-conditioned laboratory operating at 25-28°C. Flies were allowed to emerge and were fed on water and sugar for about five days, killed, identified, and counted. The percentage of fruits infested was also determined. As an example, 136 kg of mango fruits made up of 1750 fruits and 12.2 kg of *G. speciosa* fruits made up of 1289 fruits were sampled and put through the laboratory. Fruit sampling demonstrated very clearly the impact of the MAT and BAT on oriental fruit fly damage to mango fruits. Together with trapping results, fruit sampling identified the hot spots or areas where breeding populations of flies persisted.

Staging of the eradication campaigns

The eradication programme was planned in four stages. Initially, the plan was to focus on the eradication of oriental fruit fly and Pacific fruit fly; the flies attracted to methyl eugenol. The major reason was that methyl eugenol attracted flies, historically, were more responsive to the MAT technique than flies attracted to cue-lure. However, in a programme where four species are being targeted, maintaining a high degree of flexibility in operations was essential.

Stage 1: October 1998-January 1999

Stage 1 consisted of two blocking campaigns, one in late October/early November and another in December. These campaigns were less than eight weeks apart, but this was designed to avoid the pre-Christmas period. The fibreboard blocks were treated with methyl eugenol and Fipronil only at the rate of 10-12 ml of methyl eugenol plus Fipronil per block. One protein bait application was carried out in November as a preliminary field test of the newly developed protein/Fipronil bait.

Stage 2: February-October 1999

Stage 2 covered campaigns 3-7 and involved a major change to the composition of the lures in the fibreboard blocks. A mixture of cue-lure, methyl eugenol, and Fipronil was used to treat blocks. This was done to take advantage of the very low numbers of melon fly, which resulted from the low incidence of wild and cultivated cucurbit hosts due to the severe drought. As well as putting extra pressure on the population of melon fly, maintaining pressure on the seriously depleted populations of male flies of oriental fruit fly and Pacific fruit fly was also necessary. The new mixture consisted of 3 *l* of methyl eugenol plus 6 *l* of cue-lure/ethanol in a ratio of 1:9 by volume plus 28 or 36 ml of Fipronil, depending on the concentration of Fipronil in the special premix. The reasons for mixing cue-lure with ethanol was to reduce the cost using ethanol solely as a dispersant and also increase the ease of absorption of cue-lure into the fibreboard block. The amount of lure/ Fipronil per block was increased to 12-15 ml per block, to ensure that there was sufficient methyl eugenol present to remain active for eight weeks under Nauru conditions.

Stage 3: November1999-October 2000

Stage 3 covered campaigns 8-13. As Pacific fruit fly persisted in very small numbers at a limited number of locations and the percentage of traps with positive records of mango fly remained at about 30%, the decision was taken to revert to dispersing blocks treated with methyl eugenol and Fipronil only and to commence distributing blocks treated with cue-lure and Fipronil only. Mixing of methyl eugenol and Fipronil followed the system used for Stage 1. Cue-lure was diluted with ethanol in a ratio of 1:9 as in Stage 2 and mixed with 4.0 ml of Fipronil per litre. 12-15 ml of cue-lure and Fipronil was absorbed per block. The methyl eugenol treated blocks were distributed at a density of 400-700 per km², while the cue-lure treated blocks were distributed at a density of 800-1000 per km².

Stage 4: December 2000 to present

Stage 4 involved the introduction of new technology called BactroMAT M-E and BactroMAT C-L bait stations during campaign 14 on 4-8 December 2000. This involved impregnating papier-mâché discs, approximately 38 mm in diameter and 1.5 mm thick, with lure and Fipronil at Aventis CropScience in Brisbane. Relatively small numbers of BactroMAT M-E bait stations (about 2500) were distributed to ensure that Pacific fruit fly was eradicated. About 10,000 BactroMAT C-L bait stations have been distributed since December 2000. In February 2001, the use of BactroMAT M-E bait stations was terminated.

RESULTS AND DISCUSSION

Eradication of methyl eugenol-responding fruit flies

Oriental fruit fly

Oriental fruit fly occurred in very large numbers throughout the coastal area and around Buada Lagoon. For example, during October 1998, an average of 72.4-126.1 oriental fruit flies were trapped per day. At one site in the Buada Lagoon area, over 2500 oriental fruit flies were caught in one trap in a 30-hour period. Although oriental fruit flies were present in Topside, examination of fruits of Pacific almond and mango showed that there were no breeding populations as there were on the coast or in the Buada Lagoon area. Also, the drought had reduced fruiting of Pacific almond and *C. inophyllum* to a minimum. Most flies trapped on Topside were probably flies migrating from the coast or Buada Lagoon area through the area.

After two MAT campaigns using methyl eugenol/Fipronil blocks and one BAT treatment using Bactrogel protein bait, oriental fruit fly was not recorded from traps after 15 January 1999. No flies were reared from fruits after 6 December 1998. Oriental fruit fly was declared eradicated in October 1999.

Pacific fruit fly

Pacific fruit fly occurred in reasonable numbers in several areas of Nauru, considering the host range was limited. For example, in October 1998, 3.6-4.7 flies were trapped per day. Most of these flies originated from the Buada Lagoon area and the Nibok Forest and the adjacent Nauru Phosphate Corporation residential areas in the northwest and west of Nauru. Very few Pacific fruit flies were recorded on Topside, where hosts were rare. Although the initial blocking campaigns reduced fly numbers in traps to zero over the period 3 November 1998 to 10 February 1999, small numbers of flies were caught intermittently until 16 February 2000. Flies were recovered from breadfruit samples until November-December 1999. The final eradication was brought about only when methyl eugenol was separated from cue-lure in blocks in December 1999. Pacific fruit fly was declared eradicated in October 2000.

There are two possible reasons for the persistence of Pacific fruit fly beyond the time at which oriental fruit fly was last recorded in January 1999. There was evidence that Pacific fruit fly does not feed as readily on methyl eugenol as other flies attracted to this lure (e.g., the *dorsalis* complex of fruit flies). In Fiji Islands, fruit fly workers observed live Pacific fruit flies in traps that were newly baited with methyl eugenol and malathion on many occasions (A. Allwood pers. obs.). Also, combining the two lures on one block may reduce the effectiveness of each lure. The amount of methyl eugenol impregnated into each block was reduced to about 4-5 ml when both lures were impregnated into the same blocks, compared to 10-12 ml, when the block was treated with methyl eugenol alone. Previous evidence showed that too little methyl eugenol added to carriers might result in the attractant not lasting for the full eight weeks (Lloyd *et al.* 1998; Cunningham 1989; Koyoma *et al*. 1984).

Eradication of cue-lure responding fruit flies

The effectiveness of cue-lure in MAT is recognised as being less than that of methyl eugenol (Bateman 1982). Some male flies apparently achieve sexual maturity and have the opportunity to mate before their response to cue-lure is fully expressed. Consequently, while using cue-lure for MAT may significantly reduce populations of cue-lure responding flies, small residual populations are left and result in continuation of the population, unless other forms of fruit fly management are implemented. Often the use of protein bait sprays or sterile insect technique needs to be incorporated into a programme to ensure complete eradication.

Melon fly

In late October 1998, melon fly was recorded from 30% of the traps baited with cue-lure, with 2.2 flies per trap per day. By taking advantage of virtually no wild cucurbits due to a prolonged severe drought of about two years and the lack of backyard or commercial cucurbit production, the impact of a single protein bait spray application using an early formulation of Aventis's Bactrogel in November 1998 and the use of cue-lure for MAT from February 1999 was remarkable. No melon flies were recorded from the very few cucurbit samples that were taken and none were recorded in traps from 1 February 1999. Melon fly was declared eradicated in October 1999. This is the first time worldwide that melon fly has been eradicated using these methods.

Mango fly

The programme on eradication of mango fly is still operating. Mango fly occurred in all traps in Nauru, often in very large numbers, especially in areas such as Buada Lagoon and Nibok Forest on the west coast. In October 1998, 379-912 flies per day were trapped. These fly numbers were typical of mango fly in other Micronesian countries, such as in Pohnpei in Federated States of Micronesia (Leblanc and Allwood 1997). Mango (0.12-2.46 flies per fruit), guava (2.0-27.1 flies per fruit), Pacific almond (3.8-15.1 flies per fruit), and *G. speciosa* (0.2-1.7 flies per fruit) contributed to the large populations of mango flies. As a result of the MAT programme using cue-lure and Fipronil, either in combination with methyl eugenol or alone, the numbers of flies were reduced to 0.02-0.03 per trap per day by April 2000. The percent of traps with positive records of mango fly decreased from 35.9% in early January 2000 to 7.7% on 5 April 2000. Reduced numbers of mango fly were due to changing to blocks treated with cue-lure/Fipronil alone and a concerted effort in protein bait spraying using Bactrogel, especially in the Buada Lagoon and Nibok Forest areas. Unfortunately, since then, mango fly numbers have increased substantially, due mainly to reduced local commitment, ineffective distribution of blocks or BactroMAT C-L, irregular bait application, and insufficient coverage by protein bait sprays and blocks.

Quarantine preparedness

The Government of the Republic of Nauru drafted and promulgated its first Agricultural Quarantine Act to allow for protection against re-entry of produce infested with exotic fruit flies or other quarantinable pests. Training of a small corps of four quarantine officers is being done in Pohnpei under the guidance of the SPC Plant Protection (Micronesia) Project. The quarantine surveillance system of trapping is being maintained as an early warning system for Nauru. Staff are trained in emergency response procedures for exotic fruit flies and supplies are available if a response is necessary.

Benefits and lessons learnt

Nauru people now have access to a limited amount of fresh fruits (e.g., mangoes, guavas, soursop, mountain apples, and breadfruit), which are virtually free of damage by fruit flies. Public interest in growing tropical and sub-tropical fruits has been generated, resulting in a project for a small nursery for propagation of planting material of exotic fruit trees being developed by the Departments of Youth and Education. This approach is a natural flow-on from the successful eradication programme and has potential to substantially increase the availability of wholesome, fresh food for a society that has unacceptably high incidences of obesity, coronary disease, and diabetes. To improve the diets of the people by substituting even small amounts of fresh fruits may have a major impact on the health of people in Nauru.

Improving quarantine capacity in Nauru overcame a void in the quarantine chain across the Pacific and provided greater plant protection, both nationally and regionally. The eradication programme in Nauru provided the opportunity for hands-on training on fruit flies, eradication techniques for fruit flies, quarantine surveillance, and emergency response planning to cope with exotic outbreaks. Since October 1998, over 40 plant protection and quarantine staff from 18 PICTs (American Samoa, Cook Islands, Federated States of Micronesia, Fiji Islands, Guam, Kiribati, Marshall Islands, New Caledonia, Niue, Palau, PNG, Samoa, Solomon Islands, Tokelau, Tonga, Tuvalu, Vanuatu, and Wallis and Futuna), New Zealand, and the SPC Plant Protection Service spent 2-4 weeks in Nauru undergoing field training. Part of this training included the drafting of emergency response plans for the eradication of introductions of exotic fruit flies for the respective PICTs.

The major lessons learnt during this eradication exercise are that having early warning systems in place and having a well documented, and preferably tested, emergency response strategy will save an enormous amount of time and funds in the event of an incursion of an exotic pest. Also, the technologies for eradication of many fruit fly species are available, but the best technology is only as good as the technical and management commitment and support of the field operatives and the government. Premature reduction of inputs into MAT or BAT or quarantine in a fruit fly eradication programme and reduced commitment may be disastrous to the programme and also undermine the confidence in the technology. There are deficiencies in the technologies available for eradication of some fruit fly species, especially those that do not respond to either methyl eugenol or cue-lure. The deficiencies exist in not having adequate methods of eradication, but also in not having reliable quarantine surveillance systems that will allow authorities to detect incursions of pest species as early as possible.

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Man-made marinas as sheltered islands for alien marine organisms: Establishment and eradication of an alien invasive marine species

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Abstract The typical tidal range in the west and north-west areas of the Northern Territory, Australia, is 8 m. Four sheltered marinas with double lock gates have been developed to date from the Darwin Harbour estuary, or dug from the shoreline, to provide regulated environments with no tidal range. These sheltered marinas are novel environments and provide habitat islands for colonisation by invasive alien marine species. In March 1999, a fouling mussel, *Mytilopsis* sp., closely related to the freshwater zebra mussel, *Dreissena polymorpha,* was discovered in one of the marinas at densities up to 23,650/m^2. It had reached those densities in less than six months. We describe the colonisation of this and other marinas by the mussel, and the approaches taken to quarantine and eventually eradicate it. Lastly, we discuss the features that may have led to the invasion and present actions that are being taken to reduce the risk of future invasions.

Keywords invasive alien marine species; marine pests; mussels; *Mytilopsis* sp.; marinas; chlorine; copper sulphate; detergent; temperature.

INTRODUCTION

Increased population and expanded tourism in coastal regions has resulted in an increasing number of man-made structures to service the economic, residential, recreational and aesthetic desires of coastal communities. These novel physical habitats or habitat-islands (e.g. piers, breakwaters, seawalls, eutrophic and polluted areas, docks and marinas, boat hulls and ballast tanks) often support assemblages that are distinct from neighbouring communities (Glasby 1999). So long as the novel assemblage is formed from elements of local communities this is not a major concern. However, when novel physical habitats are developed in areas subject to a high influx of alien organisms, the combination could increase opportunities for invasion by alien species and a source for colonisation of adjacent established communities (cf. MacArthur and Wilson 1967).

Boat marinas in particular are novel marine habitat islands in a colonisation corridor. Since 1988 four boat marinas, closed off from adjacent waters by double lock gates, have been built in Darwin, Northern Territory, Australia. On 27 March 1999, CSIRO divers undertaking a survey for exotic species discovered huge numbers of an unidentified mussel in one of these marinas (Bax 1999; Ferguson 2000; Willan *et al*. 2000). The mussel, nominally *Mytilopsis sallei*, is a close relative of the zebra mussel *Dreissena polymorpha*, a species estimated to cost U.S raw water-dependent infrastructure USD18 million in 1995 alone (O'Neill 1996).

Based on literature reports of the environmental and infrastructure damage caused by *D. polymorpha* in the US, and by *Mytilopsis* sp. in Southeast Asia, where it is introduced (Morton 1989), the mussel was seen as a threat to water-dependent marine infrastructure around northern Australia, to a local A$40 million pearl fishery, and to the environment. Given this threat, and the apparent restricted distribution of the mussel, the Northern Territory Government determined that a fast and vigorous response was called for including, if possible, eradication of the mussel. The Northern Territory Government has a history of responding rapidly and effectively to invasions of terrestrial pests that threaten the local (and national) agricultural industries; here, they extended this experience to a marine alien invasive species.

In this paper we describe, the response by the Northern Territory Government and by Australian national agencies to control the new invasion, to reduce the risk of the species spreading in Australia, and to reduce the likelihood of future introductions of this species. We describe the successful eradication effort, discuss the lessons learned from it and further consider the conditions that contribute to invasion of these marine habitat islands.

Following Willan *et al.* (2000), we use *Mytilopsis* as the genus name. However, because there is some confusion in the literature over the species-level identification of *Mytilopsis* species in Southeast Asia (see contrasting views in Morton 1981 and Marelli and Gray 1985), and because the different species may have different environmental limits and potential impacts, we refer to the mussel as *Mytilopsis* sp. in this paper. For legislative purposes, the mussel was referred to as *Mytilopsis* (= *Congeria*) sp. Despite detailed morphological examination of the specimens from Darwin, it is still not clear which species of *Mytilopsis* invaded, which reflects the uncertain taxonomy of the genus.

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Table 1 Sequence of events associated with the eradication of *Mytilopsis* sp. in Darwin, Australia (data from Ferguson 2000).

Date	Event
27 March 1999	Massive infestations of colonising mussel found in Cullen Bay Marina
29 March	Northern Territory agencies and minister informed
30 March	Special meeting of cabinet to pass regulatory amendments and approve expenditure of funds
31 March	Emergency management team convened; three marinas quarantined to prevent further spread of *Mytilopsis* sp.; marina locks dosed with sodium hypochlorite to create sterile plug
1 April	Media and public informed
2 April	Extensive diver surveys began; list of potentially colonised vessels developed
3 April	Copper sulphate tested in Tipperary Waters Estate Marina
4 April	Chlorine treatment of Cullen Bay Marina; Vessel tracking database established (420 vessels identified as “at risk”); treatment of vessels’ internal plumbing tested
5 April	Chlorine treatment of Cullen Bay Marina continued
6 April	Chlorine treatment of Cullen Bay Marina continued; National Taskforce established; 100% kill rate in Tipperary Waters Estate Marina
7 April	Copper sulphate treatment of Cullen Bay; chlorine treatment of Frances Bay Mooring Basin; vessel cleaning protocols released; scientific sub-committee of National Taskforce established
8 April	Copper sulphate added to Frances Bay Mooring Basin; endoscopes used to check internal plumbing of vessels in Cullen Bay Marina
9 April	Further chlorine treatment of Cullen Bay Marina following heavy rain
12 April	No live mussels in Cullen Bay Marina monitoring areas; some cleaned vessels allowed to leave
16 April	Surviving mussels detected on vessels leaving Cullen Bay Marina; marinas closed again and quarantined; intensive diver surveys of marinas; National protocols formally released
17-19 April	Intensive sampling of Cullen Bay Marina detected two live and two recently-dead mussels among hundreds of thousands of dead mussels; copper sulphate added to specific sites in marina
20 April	Cullen Bay Marina reopened at high tide for limited access; resurvey of Tipperary Waters Estate Marina to confirm absence of live mussels
22 April	Resurvey of Frances Bay Mooring Basin to confirm absence of mussels
23 April	Quarantine lifted from all three marinas; marinas re-opened for normal use; monitoring continued
29 April	National Taskforce ceased operation
8 May	21 day “all-clear” issued for all three marinas. Precautionary vessel checking and treatment arrangements remained in place
July	National Taskforce for the Prevention and Management of Marine Pest Incursions established to examine all aspects of alien invasive marine species management
December 23	Taskforce report delivered to government ministers
4 January 2000	Contracts signed for development of comprehensive databases to assist future rapid responses to alien invasive marine species
22 Dec 2000	Web-based toolbox of all documented control measures used against alien invasive marine species completed and online

THE INVASION

Detection

The mussel was discovered in Cullen Bay Marina on 27 March 1999 at densities up to 23 650 individuals/m^2 (Bax 1999; Ferguson 2000; Willan *et al.* 2000)(Table 1). It was not present six months earlier in the dry season baseline survey conducted by the same divers in the same locations (C. Hewitt, CSIRO pers. comm.), indicating that the mussel has the potential for explosive population growth in these marina environments.

Several days later the mussel was also found in low numbers in a second marina, known as Tipperary Waters Estate Marina. This marina had only been recently excavated from dry land and had only six yachts berthed within it. The mussel was found on the hull of one recreational yacht that had come from Cullen Bay Marina, and on the adjacent pilings.

A third marina, Frances Bay Mooring Basin, had also received yachts that had been moored in Cullen Bay Marina. One of these was fouled with the mussel.

Characteristics of the invaded area

Darwin in the Northern Territory of Australia (Fig. 1) is an area of environmental extremes; 8 m spring tides are common, and monsoonal climate provides alternating wet and dry seasons. The extreme tides limit successful settlement of marine species due to the associated strong currents and large expanse of exposed intertidal habitat. Native species are adapted to this environment. Closed mari-

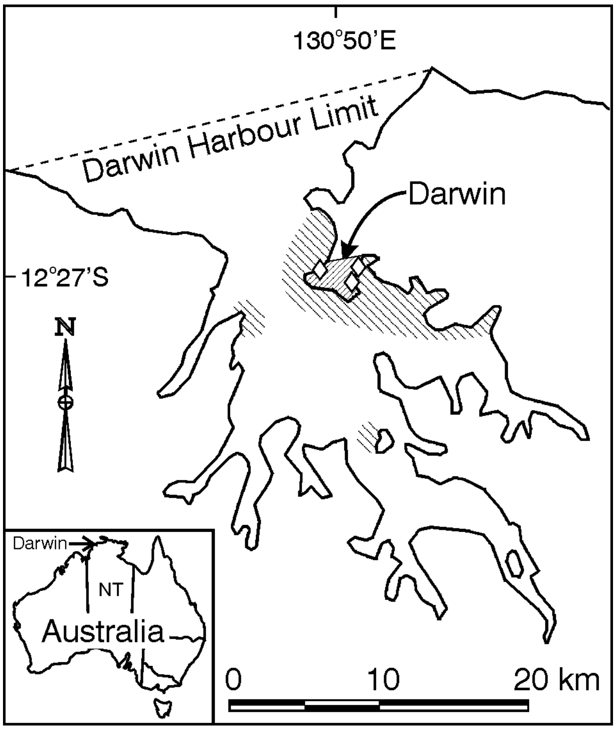

Fig. 1 Map of Darwin, indicating the three marinas (diamonds) that were treated and the areas (hatched) within Darwin Harbour where all man-made structures (wharves, oil rigs, channel markers, etc) and anchorages were surveyed by divers. Gove Harbour (600 km to the east) was also surveyed by divers.

nas prevent the large tidal excursion, and thus provide a novel habitat island that may be more readily colonised by alien invasive marine species.

At the time of the invasion there were three marinas in Darwin (Fig. 1): Frances Bay Mooring Basin, a 250 ML primarily commercial vessel marina with 83 berths that opened in 1988; Cullen Bay Marina, a 600 ML recreational marina with 135 berths that opened in March 1994; and Tipperary Waters Estate Marina, a 150 ML recreational marina with 77 berths that opened in March 1999.

Double lock gates operate to pass vessels into and out of the marinas at all stages of the tide. Depending on season and the level of flushing by the marina operators, the marina can be strongly stratified with an overlying freshwater lens up to a metre deep.

Biology of *Mytilopsis*

The two extant Dreissenidae genera (*Dreissena* and *Mytilopsis*) probably originated in the Ponto-Caspian Basin during the Eocene (Marelli and Gray 1985). The extant five to nine *Mytilopsis* 'species' occur principally in the Americas; one was introduced to north-west Europe in the late 19th century and there is another in western Africa. All species are mytiliform, byssate and epifaunal and inhabitat brackish waters. There are two tropical species, either of which could have invaded Darwin. *M. sallei* occurs naturally in the West Indies, along the Caribbean coast of Central and South America from Yucatan to Venezuela, and in southern peninsular Florida, U.S.A. *M. adamsi* (Morrison 1946) occurs in western Panama. *M. sallei* occurs naturally in a low tidal regime; *M. adamsi* occurs naturally in areas of high tidal range. Both tolerate varied environmental conditions (10-35ºC and 0-80 ppt salinity). Because of the uncertainty about which *Mytilopsis* species was present in Darwin, the wide tolerance of the genus added to concerns over the range of habitats potentially at risk.

Sometime prior to 1929, *Mytilopsis* sp. was reported from Fiji (Hertlein and Hanna 1949). Species identity is still uncertain. It may have been *M. sallei*, entering the Pacific after the Panama Canal was opened (Morton 1981), or *M. adamsi* that occurs naturally in the Pacific and extended its range with the assistance of mail steamships that plied the Pacific between Panama and Australia in the late 19th century using Fiji as a port (Marelli and Gray 1985). *Mytilopsis* has since been recorded from India, Japan, Taiwan and Hong Kong (1967, 1974, 1977 and 1980, respectively)(Morton 1981).

Ripe individuals of *M. sallei* are found all year round in the brackish water of its native range, but it has two periods of intense spawning activity, apparently stimulated by rapid drops in salinity resulting from seasonal freshwater outflow (Puyana 1995). Outside of its native range, nominal *M. sallei* favours disturbed environments, spawns twice a year and may be ambisexual (Karande and Menon 1975), or predominantly semelparous (Morton 1989). Ambiguity in the literature over the reproductive biology of invasive populations could indicate that more than one species has colonised Asian ports. Juveniles from the year's first spawning are mature within a month and contribute to the year's second spawning event (Karande and Menon 1975; Morton 1989). The one month maturity led us to assume that an infected vessel would be able to transmit *Mytilopsis* sp. 30 days after being exposed to a viable population.

PRELIMINARY STEPS

Hazard analysis

Risk assessment provides a framework to weigh the relative costs and benefits of an eradication effort (Bax *et al*. 2001). There are usually four or five stages to a risk assessment (Hayes 1998). The first stage is often to identify all potential hazards associated with a particular event; the second to quantify the risk associated with each hazard. Hazard identification serves an important role itself by providing a checklist of the hazards that need to be considered and (potentially) their relative importance. There is a wide variety of hazard identification techniques. Most of these involve 'workshops' with persons well acquainted with the area or system where the hazards are to be identified. There was insufficient time (or established protocols) to use a workshop approach in this case. Instead, scientists from the CSIRO Centre for Research Into

Table 2 Potential vectors for *Mytilopsis* sp. in Cullen Bay.

Vector	Larvae		Adults	
	Ballast Water	Other Water	External Fouling	Internal Fouling
International and domestic yachts (long & short term residents)	X	X	X	X
Quarantine vessel, dive boats, naval vessels, Fisheries Protection vessel[1]		X	X	X
Ferries	X	X	X	X
Recreational craft (e.g. dinghies, jet-skis, outboards, etc.)		X	X	X
Fishing gear & nets			X	
Buoys/traps/floats[2]			X	X
Loch water		X		
Bay water or substrate samples (e.g. for aquaria, bait)		X		
Flotsam and jetsam		X	X	
Fauna (e.g. birds, crustacea)			X	
Pipe reverse flow (e.g. stormwater overflow, sewage)		X	X	X

[1] These vessels do not hold substantial quantities of ballast water because they don't load and unload large quantities of cargo (some naval vessels might but not the ones in Darwin at the time).
[2] Damaged buoys may hold small quantities of water. Floats are usually porous to some degree hence they may hold water but this is not a viable vector for larvae.

Marine Pests (K. H., R. T., N. B. and Chad Hewitt) conducted an initial hazard analysis for *Mytilopsis* sp. in Cullen Bay Marina.

Egg/sperm and larvae were treated as larvae, while juveniles and adults were treated as adults to reduce the number of hazard analyses required with no perceived loss of hazard identification. A lack of information on larval settlement preferences led us to assume that larvae would act passively in the water column with no settlement preferences, and that juveniles and adults would settle permanently and mature rapidly. These assumptions tend to overestimate potential hazards and are therefore conservative.

Four main ways in which mussels could leave the marinas were identified: in ballast water; other water (e.g. bilge water, anchor well water, etc.); external fouling on the exposed hull, and internal fouling in pipes; and inlets leading off the hull (Table 2). Simple 'fault trees' were constructed for adults carried as external or internal fouling and for larvae carried in ballast or other water (Tables 3 and 4). Hazard management options were then developed (Table 5).

A hazard analysis was also carried out on vessels that had potentially been in an infested area. Four risk categories were identified for areas, and vessels were assigned the risk level of the area they had entered. The hazard analysis suggests that the pelagic larval life-stages of *Mytilopsis* sp. are the most "infectious" and therefore the most likely means of transmission of the organism beyond the infestation. A simple qualitative risk assessment was therefore implemented along the following lines:

Confirmed high risk areas: those areas where spawning had been shown to have occurred; Cullen Bay and Tipperary Waters Estate marinas only.

High risk areas: those areas exposed to an extant population of *Mytilopsis* sp. (i.e. on an infected vessel) and where there had been insufficient in-water surveys or larval/post larval collections (see below) to determine whether spawning had occurred.

Medium risk areas: those areas where a reproductive population of *Mytilopsis* sp. was known to have been (ie on an infected vessel) but, either the source of infection posed a medium risk (i.e. a vessel exposed in a another medium risk area), or extensive and weekly in-water surveys or surveys using larval settlement plates detected no indications of larval settlement.

Low risk areas: those areas that had either been treated or had had two in-water surveys one month apart with no detection of juvenile or adult *Mytilopsis* sp., were subsequently monitored monthly for post larval settlement, and had not received untreated vessels from medium or high risk areas since treatment or completion of surveys.

These categories were used to set priorities for interdicting and treating potentially infested vessels.

Surveys

Twenty-eight divers, supported by surface teams to withstand the strong currents and to keep watch for crocodiles, conducted systematic surveys of all apparently suitable habitats for *Mytilopsis* sp. in the three marinas, around Darwin Harbour and as far afield as Gove Harbour; a harbour with a 1 m tidal range, frequented by visiting international yachts, served by ferry from Cullen Bay Marina, and therefore deemed to be a high risk area (Ferguson 2000). Barges, oil rigs, wharf piles, the naval base and sewage and storm water drains in the three marinas were among the habitats inspected.

Table 3 Hazard analysis for *Mytilopsis* sp. adults in external and internal fouling.

ENDPOINT	NECESSARY EVENTS	
Escape of adult *Mytilopsis* on external and internal fouling vectors	1. Adults remain viable on exit from Cullen Bay.	a. Oxygen remains within tolerable limit b. Sufficient moisture to prevent desiccation
	2a Vector infected with adults in Cullen Bay OR	a. Vector in Cullen Bay during settlement of larvae following period of spawning b. Larvae settle on vector
	2b Vector picks up dislodged adults	a. Adults dislodged b. Adults re-attach to vector
	3. Vector leaves Cullen Bay	a. Vessel movement out of the bay (international and domestic yachts, quarantine vessel, dive boats, ferries, naval vessels, fisheries protection vessel, recreational craft) b. Movement of gear out of bay (outboard motors, buoys/ pots/ floats, fishing gear and nets) c. Other material movement out of the bay (bay substrate, flotsam, other fauna)

Table 4 Hazard analysis for *Mytilopsis* sp. larvae in ballast and other water.

ENDPOINT	NECESSARY EVENTS	
Escape of larvae *Mytilopsis* in ballast water and other water	1. Larvae remain viable on exit from Cullen Bay	a. Salinity remains within tolerable limits b. Temperature remains within tolerable limits c. Oxygen remains within tolerable limits d. Sufficient food sources e. Sufficient moisture to prevent desiccation
	2. Vector infected with larvae	a. Vector in Cullen Bay during period of spawning and prior to settlement
	3. Vector leaves Cullen Bay	a. Ballast water exit from Cullen Bay (international and domestic yachts, ferries) b. Other vessel water exit from Cullen Bay (bilges and other sea water systems of yachts, quarantine vessel, dive boats, ferries, naval vessels, fishing protection vessel and recreational craft) c. Other water exit from bay (leakage from lock, samples of bay water, outboard motors, flotsam)

Table 5 Hazard management options for *Mytilopsis* sp. in Cullen Bay Marina.

POTENTIAL VECTOR	SUGGESTED MANAGEMENT
International and domestic yachts (long & short term residents)	Clean external submerged surfaces Treat internal seawater systems Treat ballast (or residual ballast in empty tanks) Remove domestic yachts from Cullen Bay once cleaned
Quarantine vessel, dive boats, naval vessels, Fisheries protection vessel	Clean external submerged surfaces Treat internal seawater systems
Ferries	Clean external submerged surfaces Treat internal sea-water systems Treat ballast (or residual ballast in empty tanks)
Recreational craft (e.g. dinghies, jet-skis, outboard motors, etc.)	Clean external submerged surfaces Clean and dry internal seawater systems Educate users and repairers on risks
Fishing gear & nets	Clean and dry on removal from bay Educate users on risks
Buoys/pots/floats	Clean and dry Ban removal from bay Educate users on risks
Loch water	Maintain positive pressure into bay Treat or prevent escape of lock water
Bay water or substrate (e.g. for aquaria, bait)	Educate users on risks
Flotsam and jetsam	Dry prior to onshore disposal Prevent escape via lock
Fauna (e.g. birds, crustacea)	Verify the importance of this vector prior to management
Pipe reverse flow (e.g. stormwater overflow, sewage)	Clean Ensure positive pressure into bay

Marina operators record vessels entering and leaving the marinas as they pass through the lock gates. All vessels found to have been in the marinas during the time that *Mytilopsis* sp. was present and capable of spawning (taken to be one month after the August 1998 survey when no mussels were detected) were tracked and located. Survey protocols to inspect vessels were developed in conjunction with boatyard operators and the fishing industry. Particular attention was given to the hull surface, ropes, chains, anchors, seawater inlets and internal water systems. Approximately 250 vessels were inspected by divers (Ferguson 2000). Divers used 3 m and 22 m long endoscopes to survey the interior spaces (e.g. water intakes and outlets).

Laboratory trials of treatment options

Chlorine and chlorine dioxide are frequently used to remove *D. polymorpha* from water-based infrastructure in the US (Boelman *et al*. 1997), and it seemed likely that they would be suitable to remove *Mytilopsis* sp. from the marinas. The relatively easy availability of chlorine (as hypochlorite), which is used to clean swimming pools, made this an attractive option. Chlorine dioxide was also available, and in theory should have been more effective than hypochlorite, so this was a second option. However, *D. polymorpha* is a freshwater species, and there was no guarantee that chlorine would be equally effective against the marine *Mytilopsis* sp. Therefore four additional treatments were tested: copper sulphate; a patented organic copper complex; hot water; and detergent. Non-oxidising chemicals (e.g. quaternary ammonium compounds), reported to kill the zebra mussel in the U.S, were not tested as sufficient quantities for treatment could not be located in Australia.

All tests were conducted in triplicate on Cullen Bay Marina mussels held in 2 *l* glass beakers of Cullen Bay Marina water at the Northern Territory University. The salinity was 18 parts per thousand (ppt), pH 8.1, turbidity 2-3 nephelometric turbidity units; dissolved oxygen 90-100%; temperature 29-33°C (ambient). Salinity was lower than seawater as it was at the end of the wet season and the locked marinas had significant freshwater inflow. Beakers were covered with a watch glass cover to reduce evaporation. Mussels were obtained fresh from the marinas and were not fed. Approximately 30 individuals with maximum shell length of 1 to 1.5 cm were used in each beaker. The LT_{100} (time to achieve 100% mortality) was determined for each treatment. Death was determined as gaping shells unresponsive to touch. The LT_{100} was used in preference to the LT_{50}, more commonly used in toxicity trials, as we required a treatment that would kill all of the mussels. Chemical concentrations were checked twice a day (colorimetric method for chlorine; ICP Mass spectrometry for copper), and additional chemicals added if the concentration had dropped below the test level. Laboratory conditions were not ideal standardised conditions because they were hastily set up to identify an effective treatment within days, however they were thought to more closely represent the conditions that would occur when treating a marina.

Calcium hypochlorite

At least twice-daily measurement of chlorine concentration showed that the nominal concentration in the beakers were difficult to maintain – in fact 12 hours after adjustment of concentrations to 12 or 24 mg/*l*, concentrations weer approaching <1 in the day and 5 mg/*l* at night. This was expected as chlorine is unstable in water and exposure to light and elevated temperatures accelerates the reduction in chlorine concentration. Chlorine concentrations were adjusted after each measurement to maintain the test concentration. The time to 100% death (LT_{100}) ranged from >290 hours at a nominal concentration of 0.0, 0.6, 1.2 and 6.0 (mg/*l*) to 111 hours at a nominal concentration of 12.0 (mg/*l*) and 90 hours at a nominal concentration of 24.0 (mg/*l*). From our experiments, we thought it likely to prove difficult to maintain these concentrations in the infested marinas.

Chlorine dioxide

From literature reports for zebra mussels, we expected chlorine dioxide to be more effective at killing the mussels than hypochlorite. However the protocols provided by the manufacturer for activating the stabilised chlorine dioxide solution did not work – hydrochloric acid activation resulted in the complete loss of chlorine dioxide in <15 hours and citric acid activation did not activate the chlorine dioxide at all. All further tests with chlorine dioxide were abandoned as we did not have the time to work out the correct activation procedure.

Copper sulphate

Copper sulphate was tested on the basis of its common use to kill invertebrates in aquaculture operations. Copper was added as copper sulphate to give a nominal (and subsequently measured) concentration of 1 mg of copper/*l*. In seawater (35ppt) the concentration of Cu is also controlled by copper hydroxide solubility and the saturated concentration in seawater is 2 µM (0.126 mg/*l*). However due to the low salinity water in the marina a concentration of 1 mg/*l* could be maintained. The LT_{100} for copper sulphate was 38 hours.

Copper organic complex

Organic complexes of copper are generally considered to be non-toxic or have lower toxicity than free copper ions. This was confirmed in our trials, which were ended after 48 hours when mussels were still alive in the two treatments (0.5 and 1.0 mg/l).

Combined calcium hypochlorite/copper sulphate

This experiment, designed to simulate possible field trials, used a nominal chlorine concentration of 12mg/*l* chlorine (see notes under calcium hypochlorite experiments) for either 24 or 48 hours followed by addition of 1 mg/*l* Cu. Both treatments resulted in a LT_{100} of 96 hours.

Detergents

Detergent was tested following observations of its lethal effect on marine life in aquaria, and with a view to using it to clear the internal plumbing of vessels. Domestic detergent (1% v/v) in seawater of 13 and 33 ppt salinity gave LT_{100}s of 24 hours. Industrial detergent (Conquest, 1% v/v) in 19 and 33 ppt salinity seawater gave LT_{100}s of 7 hours.

Temperature

Mussels were tested in beakers placed in temperature controlled water baths and held at 40, 50 and 60°C. The LT_{100}s were >120, 30 and 30 minutes, respectively.

THE RESPONSE

Legislative powers and coordination

The Northern Territory Government took a 'whole of government' approach involving all departments, coordinated by the Assistant Chief for Police, Fire and Emergency Services and overseen by the relevant minister. The Minister and the Chief Executive Officer of the Department of Primary Industry and Fisheries were briefed on 29 March 1999, two days after the mussel was first observed (Table 1). The Northern Territory cabinet was informed at the same time. The following day, a special meeting of cabinet was held to amend the Northern Territory *Fisheries Act 1988* to list *Mytilopsis (=Congeria)* sp. as an aquatic pest; to give aquatic pests the same status as diseases and contaminated fish; to decree that restricted areas apply to aquatic pests; and to declare the marinas to be restricted areas and prohibit the movement of aquatic life from these areas (Ferguson 2000). The amendments to the Fisheries Act 1988 were gazetted the following day (31 March 1999) and the three marinas quarantined using a combination of this act and the *Quarantine Act 1908*.

A Northern Territory Taskforce was set up with units responsible for media, vessel tracking, emergency services, health, diving/survey, eradication, biology and treatment. Seven Northern Territory and two national organisations were involved in the local eradication (see acknowledgements for full listing); a further seven national organisations and the States were involved in the national response.

Twice daily meetings of the Taskforce kept all members informed and enabled the rapid identification of priorities and the necessary resources to meet them.

Media issues and community response

A media team was established at the start of the response, using experts from the Northern Territory Police, Fire and Emergency Services accustomed to dealing with disaster response. Immediately the marinas were quarantined, there was pressure from concerned residents, vessel owners, tourism operators and marina businesses to limit the response, especially as the quarantine went into effect just before the Easter holidays. Conversely, local aquaculture businesses and conservation groups wanted assurance that the quarantine and treatments be sufficient to ensure that the eradication was successful. The media group kept the local community, the nation, and international interests informed with daily press releases; regular community fliers; community meetings at Cullen Bay Marina involving the Northern Territory Primary Industry and Fisheries Minister and senior Taskforce members; a public telephone hotline; and a web site.

The work of the media team was critical in gaining community and stakeholder support. The team ensured that consistent and informative messages were provided on Taskforce activities. Press releases and media events were managed so that a new topic was presented each day by authoritative figures and informed scientists, reducing the need for the media to look elsewhere for the day's story. The involvement of national (CSIRO's Centre for Research on Introduced Marine Pests) as well as local scientists in the eradication programme and media interviews, facilitated community and stakeholder engagement in the issue by emphasising that all available resources were being accessed.

Quarantine

Colonised marinas were quarantined on 1 April 1999; five days after the mussels were first observed in Cullen Bay Marina and three days after their identification was confirmed. No vessels inside colonised marinas were permitted to leave until the marinas were declared free from the mussel. Once the three marinas had been quarantined, locally available sodium hypochlorite was added to the short (<40 m) channel between the two lock gates separating each marina from the ocean. This quarantine was used to prevent larval *Mytilopsis* sp. from leaving the marinas alive and to kill any *Mytilopsis* sp. occurring in the channel.

Chemical eradication

Marinas

Treatments for the marinas proper needed careful consideration, due to both their size and usage. The largest, Cullen Bay Marina, had the highest densities of the mussel, up to 23,650/m^2 (Willan *et al.* 2000), compared to 6/m^2 (Ferguson 2000) in Tipperary Waters Estate Marina. None were sighted in Frances Bay Mooring Basin (although an infested yacht that had recently been in Cullen Bay Marina was found in the Basin). Treatment was therefore focused on the Cullen Bay Marina and the vessels still moored there. The smaller Tipperary Waters Estate Marina was used as a field experimental site.

Based on US experience with the freshwater *D. polymorpha* (Boelman *et al.* 1997), chlorine was the preferred option for chemical eradication. It was estimated that several hundred tonnes of chlorine, in the form of liquid sodium hypochlorite, would be needed to raise the larger 600 ML Cullen Bay Marina to 10 ppm free chlorine, and hundreds of tonnes of sodium hypochlorite were shipped to Darwin from chemical plants around Australia. Estimates were of necessity imprecise as we could not accurately estimate the likely loss of chlorine through evaporation and being bound to organic matter. Large pumps were used to aerate the 12 hectares of water in Cullen Bay to raise the oxygen levels, break up the stratification of the salt and freshwater layers, and disperse the chlorine. The first load of sodium hypochlorite was added to Cullen Bay Marina on 4 April 1999, three days after it was quarantined. Concentrations were tested daily thereafter and additional sodium hypochlorite added as necessary to keep concentrations at about 10 mg/*l*.

Following early laboratory data showing the efficacy of copper sulphate, 0.5 tonne was added to 150 ML Tipperary Waters Estate Marina on 3 April 1999 – producing a maximum final 0.45 µm filtered, concentration of 0.8 mg/*l* and a total copper concentration of 1.5 mg/*l* (Parry *et al.* 1999). Measurements at top and bottom of the water column at five sites throughout the marina showed the copper was well mixed with uniform concentrations throughout. The concentration dropped rapidly after significant freshwater inflow to the marina on 8 April with the filtered and total Cu concentrations stabilising between 0.2 and 0.3 mg/l for the following two months. No sodium hypochlorite was added to this marina. A complete census of Tipperary Waters Estate Marina conducted by divers on 6 April 1999, found only dead *Mytilopsis* sp. – this was three days after the addition of copper sulphate. All mussels attached underneath foam panels floating on the surface were dead.

Meanwhile, daily, non-quantitative observations of the mussels in Cullen Bay Marina, and quantitative monitoring of caged mussels, showed that many mussels were surviving the chlorine treatment. Additionally, death rates in the laboratory were not as high as had been hoped for (see section on laboratory results). Laboratory tests and the trial of copper sulphate in Tipperary Waters Estate Marina indicated that copper sulphate was a more effective method to kill *Mytilopsis* sp. than chlorine in these marinas. Copper sulphate was subsequently used in conjunction with chlorine at Cullen Bay and Frances Bay marinas; chlorine's role was seen as primarily to reduce the organic load so that more free copper would be available in the water column. Powdered copper sulphate was added to the water at the aeration pumps to aid its dissolution.

In total, 187 tonnes of liquid sodium hypochlorite and 7.5 tonnes of copper sulphate were added to the three marinas over two weeks (cited in Willan 2000). Sodium hypochlorite was added to Cullen Bay Marina and Frances Bay Mooring Basin prior to addition of copper sulphate. There was no similar pre-treatment of the Tipperary Waters Estate Marina. A maximum final, dissolved (<0.45 µm) concentration of copper of 0.8 mg/*l* was reached in Tipperary Waters Estate and Frances Bay Mooring Basin, while the concentration in Cullen Bay reached 0.5 mg/*l* (Parry *et al.* 1999). The copper concentrations in all three marinas remained uniform throughout the water column, with no stratification observed due to the mixing with aeration pumps and the running of vessels' engines in the marinas during the treatment. The maximum concentrations were only maintained for approximately two days before the levels began to decline due to various precipitation, adsorption and complexation processes in the water column.

Local vessels

Vessels inside the marina were treated at the same time as the marina – external hulls were treated by the surrounding water. Interior plumbing on all vessels was treated by running the relevant pumps or engines and adding copper sulphate solution or detergent to pipes with standing water. Where the owner was unavailable, Fisheries Officers entered the vessels and performed the necessary tasks.

Colonised vessels outside the marinas were either hauled out and cleaned at the nearest facility that had been approved to clean vessels and dispose of the mussels without risking further colonisation, or taken into one of the three colonised marinas for treatment. Two moorages outside the marinas, but still in the Darwin area, had received a total of six colonised vessels. Diver surveys detected no mussel populations at these moorages.

National taskforce

A national taskforce was set up and coordinated by the Commonwealth Department of Agriculture, Forestry and Fisheries – Australia. A scientific sub-committee was set up to develop national protocols for treatment of vessels and for anchorages at potential risk of secondary infestation, which included all those across northern Australia between Fremantle, Western Australia and Sydney, NSW. The sub-committee took an epidemiological approach in developing the protocols. *Mytilopsis* sp. was treated as an infectious disease with an incubation period of 30 days – this period being the minimum reported time for this mussel to become reproductive post-settlement (Karande and Menon 1975; Morton 1989). Any area or vessel which came into contact with an infested vessel after the 30-day incubation period was assigned the same level of infestation risk as the original vessel. An exposed vessel or area was considered infested until proven otherwise.

All told, 223 vessels were within the three marinas and another 197 had left the contaminated areas and put to sea during the time that the marinas were exposed to the mussel (Ferguson 2000). It was therefore urgent that exposed vessels and the areas that they had visited be identified, surveyed and, if necessary, treated to prevent the further spread of this mussel. A database was established by Northern Territory Police and the Australian Quarantine and Import Service to track vessels that had been exposed to *Mytilopsis* sp. but had left the marinas. This database grew to include information on the infection status and whereabouts of 743 potentially-exposed and exposed vessels.

Invoking this emergency action exposed numerous problems in tracking small vessels, and especially recreational yachts, which has since been addressed by a national marine pest task force (SCC/SCFA 1999). When located, the vessels were either examined in the water by divers or removed from the water for examination under a mixture of State and Commonwealth legislation, that was sometimes found insufficient to allow the preferred treatment options. Where there were no safe local facilities to inspect a potentially-contaminated vessel (e.g. the Cocos Keeling Islands off Western Australia), the contaminated vessel was kept away from shore and freshwater influence.

Fifty seven fishing vessels had left the Frances Bay Mooring Basin shortly before it was quarantined. These 57 vessels were part of the 137-vessel Northern Prawn Trawl fleet that would disperse throughout northern Australia at the end of the fishing season. The vessels come into contact around motherships, providing a serious risk of secondary exposure to *Mytilopsis* sp.. Recalling all exposed vessels to port for treatment (in the 30 days incubation period) was not acceptable to industry during the limited prawn season. Instead the Australian Fisheries Management Authority contacted all vessels at sea requiring that they stay at sea until all the 57 exposed vessels had been determined to be clean. Divers surveyed the exposed vessels at sea and declared them clean. Individual cases of exposed vessels returning to shore for mechanical or medical emergencies were dealt with on a case by case basis that minimised further exposure of the coastline to the mussel.

Monitoring

Mytilopsis sp. was only found in two marinas – Cullen Bay Marina and Tipperary Waters Estate Marina – and on vessels originating from Cullen Bay Marina. Three separate diver surveys of Frances Bay Mooring Basin failed to find *Mytilopsis* sp. on either wharf pilings or on vessel hulls. This marina was treated as a precautionary measure because a vessel with mature *Mytilopsis* sp. had been found there and was cleaned on the hard standing area.

Divers, trained to dive in industrial situations, monitored 20 locations within Cullen Bay Marina to assess the efficacy of the chemical treatments as part of the marina clearance and re-opening process. All available habitat was searched in these 20 areas, including storm drains, the inside of debris, etc.

In addition, mussels were suspended in cages at up to three depths (1, 2 and 3 m as available) at 10 locations in Cullen Bay Marina to monitor quantitatively the efficacy of the chemical treatments. Each cage contained approximately 100 mussels. The cages were removed twice daily and the

Table 6 Total number of live, and percentage dead, *Mytilopsis* sp. held in 19 baskets at 10 locations and three depths (1, 2 and 3 m below surface as available) in Cullen Bay Marina. The first addition of chlorine to the marina was on 4 April (Day 0). Copper sulphate was first added on 7 April (Day 3).

	Day								
	-2	-1	0	1	2	3	4	5	6
Number	2149	2149	2149	2146	1767	1297	863	351	0
Average % dead	0	0	0	0	17	38	59	84	100
Min % dead	0	0	0	0	0	4	9	68	100
Max % dead	0	0	0	1	44	68	85	95	100

condition of the mussels assayed for responsiveness to physical probing.

The first deaths in the mussel cages occurred one day after the addition of hypochlorite, but mortality rates varied widely between cages (Table 6). On average, 38% of caged mussels were dead three days after the first addition of hypochlorite, at which time copper sulphate was added. Six days after the first addition of hypochlorite, and three days after the addition of copper sulphate, all mussels in all cages were dead.

Diver surveys in Cullen Bay Marina found no live mussels on 12 April, eight days after the first addition of chlorine and five days after the addition of copper sulphate. However, on 16 April, four days after no live mussels were found at the monitoring sites, live mussels were observed on a vessel slipped for maintenance after leaving Cullen Bay Marina. Subsequent diver surveys (17-19 April) found two live and two recently dead (shell open but flesh not decomposed) mussels among the several hundred thousand dead mussels collected from the 20 locations around the marina and inspected by hand. The areas where the two live mussels were found were treated with additional copper sulphate. No further live mussels were subsequently found in any marina or on any vessel.

CONFIRMING THE ERADICATION

The area immediately outside the infected marinas, and moorages and ports to which high risk vessels had been tracked (sometimes in other states), were monitored for settling mussels using larval settlement plates for up to 12 months after the chemical treatment of Darwin's marinas. Larval settlement plates were used because the planktonic larval distribution of the mussel would distribute the larval mussels broadly, increasing the chances any reproductive population would be detected.

No juvenile *Mytilopsis* sp. were found on larval settlement plates inside or outside of the marinas in Darwin or at the major ports catering for recreational yachts in Queensland or northern Western Australia. After 12 months the eradication of this invasion of *Mytilopsis* sp. was considered complete.

DISCUSSION

The rapid response to, and subsequent eradication of, *Mytilopsis* sp. by the Northern Territory Government assisted by national agencies was not only a salutary lesson for Australia on the dangers of invasive marine alien species entering tropical Australia, but also the first demonstration that successful action against invasive marine alien species was possible. The eradication operation directly involved over 280 personnel and cost in excess of A$2.2 million, excluding personnel costs. The cost was considered cheap in the light of potential damage by *Mytilopsis* sp. to tropical Australian marine industries and the environment. The chemically-treated marinas were artificial environments already polluted from maritime activities and the temporary loss of their fauna was seen as inconsequential in comparison to the threat to the northern Australian coast.

There are several lessons to be learned from this exercise.

Planning

There may be little time to respond to the invasion of a marine alien species; options available to eradicate or control a marine invader rapidly diminish over time if the invader spreads to additional areas. In a maximum of six months (since the first survey of the marinas, at which time *Mytilopsis* sp. was not present in detectable numbers) *Mytilopsis* sp. went from a presumed single population on a visiting vessel to colonising two marinas and reaching densities of up to 23,650 individuals/m^2. If the invasion had not been detected quickly, it is likely that *Mytilopsis* sp. would have established viable populations outside the closed marina environment; our observations at other tropical yacht anchorages indicate many appear to be well suited to the species. Underwater gas arc welders, temporary covering with gravel or sand, and temporary covering with plastic (containing biocides) were proposed as methods to heat, smother, or poison the *Mytilopsis* sp., respectively, if they were discovered outside a closed marina. However, none of these techniques would have been suitable for a dispersed population in an open environment. This problem was understood very early in the programme, which prompted action to ensure that any spread of *Mytilopsis* sp. was detected early, that invaded areas were quarantined immediately, and that eradication proceeded soon thereafter.

Local and national coordination was essential for a comprehensive response, but in practice had to be established quickly and in an *ad hoc* fashion. The system worked because everyone understood the scale of the problem, and an effort was made to ensure that all relevant groups had the option of participating. There were difficulties finding appropriate State (Territory) or Commonwealth legislation to support the quarantine and treatment of privately owned marinas and vessels. The Northern Territory Government had to amend their legislation to give them the power to respond. Amendments occurred within three days of the outbreak being detected – even so there was confusion over the legislative powers available, who could exercise them and who they could be delegated to. The issues of liability coverage for officers and compensation to owners were never completely resolved.

Subsequent to *Mytilopsis*, two national ministerial councils jointly established a National Taskforce on the Prevention of Marine Pest Invasions (SCC/SCFA 1999) to address the structural inadequacies of the *ad hoc* system. Immediate recommendations that have since been enacted include: establishment of a National Introduced Marine Pests Coordination Group to oversee development of interim arrangements and to develop long-term response options; establishment of a Consultative Committee on Introduced Marine Pest Emergencies to provide coordination of rapid responses in the event of a new alien invasive marine species emergency; development of information systems to speed response to new invasions; clarification and updating of legal powers for responding to alien invasive marine species; and establishment of cost-sharing arrangements between the States and the Commonwealth to fund emergency response to future alien invasive marine species. More complex issues, such as the development of a system to track small vessels, are still being considered.

Risk assessment, when implemented properly, is rigorous and systematic but also time-consuming. In this instance of rapid response, there was insufficient time for a complete risk assessment, but the hazard assessment proved useful in identifying major threats to a successful response. An approach, termed 'Infection Modes and Effects Analysis (IMEA)' has been specifically developed to identify and rank the hazards associated with small craft as potential vectors of alien invasive marine species (Hayes in press). A similar process is needed to identify and rank the hazards associated with the outbreak of an alien invasive marine species in a new area. In this instance the hazard assessment identified particular habitats (e.g. storm water drains and internal plumbing on vessels) that were subsequently targeted by the Northern Territory Government during the chemical eradication.

Rapid access to information is a necessity in responding rapidly. This is true both for the scientists and managers involved, and for the community, to ensure their support for the action. With regard to technical details, we were able to access and distribute information relevant to *Mytilopsis* sp. by querying the Sea Grants National Aquatic Nuisance Species Clearinghouse for information on eradication options for the zebra mussel. The rapid response to our query was extraordinarily valuable in rapidly assessing practical options. Subsequently, Australia has recognised the need for easy access to information on a variety of potential invasive marine species, and is about to launch an online National Introduced Marine Pests Information System to provide similar information for alien invasive marine species currently in, or likely to arrive in, Australia. A web-based toolbox of control options has already been developed. This toolbox provides a readily available source for all documented control options for the different taxa, contacts, suppliers and legal restrictions on their use (Bax and McEnnulty 2001).

The success of the eradication programme was due in part to strong community support for the effort. This was facilitated by having a full-time public relations team assigned to the problem, which ensured that media and relevant stakeholders were provided updated information daily and by ensuring that public statements regarding the infestation and eradication effort were handled by only a few designated spokespeople. Public acceptance of the effort was also facilitated by having the response action well embedded in science. Although the rapid and in some respects non-rigorous nature of the experiments to develop effective chemical treatments was clearly explained to the public, the fact that this effort was guided by a substantial literature on zebra mussel control efforts and was undertaken by a team consisting of both local (NT University and government) and national (CSIRO's Centre for Research on Introduced Marine Pests) experts lent the effort essential credibility.

This public credibility also depended, in part, on good luck – the detection of the mussels in closed marinas where chemical treatments were an option. If we are to extend the success of this eradication to include future invasions of alien marine species in more open environments, we will need to expand available treatment options to include: more specific biocides (e.g. ones that would only affect molluscs); engineering developments that would restrict the action of chosen control options to the target area; and failing that, highly-specific biological control agents that can act over a wide area.

Secondary Exposure

The potential for the Darwin eradication attempt to spiral out of control was the greatest through secondary exposure – one contaminated vessel entering an uncontaminated marina (or fishery) and contaminating tens of other vessels that subsequently dispersed to contaminate new vessels and areas. Secondary contamination of a vessel (in this case a submarine) exposed to spawning mussels (*Mytilus galloprovincialis*) on an unsuccessfully cleaned vessel (the battleship *USS Missouri*) has been observed (Carlton 2001). Secondary contamination was a real risk in the Darwin exercise.

The 57 potentially-exposed fishing vessels that left Frances Bay Mooring Basin days before the quarantine was announced, joined the 137-vessel Northern Prawn Fleet, serviced by motherships, round which the vessels congregate to unload their catch. The 137 vessels disperse to ports throughout northern Australia once the season is over. If these vessels had carried *Mytilopsis* sp. to their home ports, the goal to eradicate this mussel from Australia would have been made much harder – probably impossibly so. Because the fishing vessels carry satellite-linked vessel monitoring systems and can be contacted by the Australian Fisheries Management Authority, the 57 vessels could be contacted and surveyed. Fortunately, no contaminated vessels were found and the risk was never realised. Tracking domestic and international recreational vessels through ports throughout mainland Australia and Australia's offshore islands was not as easy. Again we were fortunate that the vessels that were tracked and surveyed were found to be uncontaminated.

Effective quarantine is a prerequisite for an eradication exercise. If *Mytilopis* sp. had colonised a marina without lock gates, where vessel names were not recorded as they passed out of the marina, then tracking and locating exposed vessels would have been impossible.

Prevention

The tidal regime in Darwin is extreme, with 8 m spring tides. Extreme tidal variation does not easily allow stable marina platforms for mooring boats and so all marinas are closed off to some extent from the sea by double lock gates, that are opened for boat passage as necessary, and to flush entrained water on an irregular basis. Depending on the degree of flushing, there can be strong stratification in the marinas during summer monsoonal rains with a 1 m thick lens of freshwater overlying marine water. Many marine organisms that have colonised the upper few metres of underwater structures during the dry season will die when exposed to continuous freshwater during the wet season, unless there is adequate flushing of the marina. New habitat will be exposed to recolonisation by local organisms and new colonisation by exotic organisms brought in on (or in) boats passing through on their way from exotic ports.

Marinas, especially closed marinas such as those in Darwin, can act as marine habitat islands. They provide novel habitat and are situated in the middle of increasingly active transport corridors for marine organisms. In 1967, MacArthur and Wilson introduced the still influential equilibrium theory of island biogeography. This addresses the question of whether the number of new invasions to an area is primarily a function of the size of the island (habitat) or the extent of immigration. To rephrase this in the context of marine habitat islands such as marinas, the question becomes whether the number of new invasions is primarily a function of the size or types of habitat or a function of the frequency of inoculation. Programmes aimed at reducing the risk of marine invasions have in general not taken account of this ongoing debate. Instead it is assumed that reducing the number of new invasions to an area is of paramount importance, and major efforts both nationally and internationally have been directed at reducing the entry of alien invasive marine species – especially in ballast water. *Mytilopsis* sp. has been detected on three foreign fishing vessels and two visiting international recreational vessels since the eradication was completed. It seems inconceivable that the Cullen Bay invasion was the first instance of *Mytilopsis* sp. arriving in Darwin, yet the most likely scenario for this invasion is that it developed from a single inoculation in a closed marina. It subsequently spread to a second closed marina, but repeated diver surveys failed to find any populations outside of these marinas. This suggests that reducing the risk of marine invasions may require management of the receiving environment in addition to reducing the frequency of inoculation.

Reducing the risk of future invasions

Australia was lucky that this first successful establishment of *Mytilopsis* sp. in Australian waters was in closed marinas. If it had occurred in the open waters of Darwin Harbour or other Australian ports, containment would have been much more difficult, perhaps impossible, depending on the extent of its spread when detected. But "chance favours only the prepared mind" (Pasteur), and we were fortunate that the Northern Territory Government was experienced in the rapid and vigorous eradication of terrestrial pests and was able to transfer that experience to the marine environment.

In preparation for future events, the Aquatic Pest Management (APM) unit was established within the Northern Territory Government. APM completed the 12 month monitoring programme for *Mytilopsis* sp. and developed and implemented protocols that will reduce the risk of a second establishment of an invasive alien marine species in Northern Territory waters.

Consultation with local stakeholders recognised the four Darwin marinas as high-risk areas, and visiting international vessels as high-risk vectors. Since the eradication, all international vessels wishing to enter Darwin marinas are now inspected and treated prior to being issued clearance certificates. Entry to the marinas is prohibited without a clearance certificate.

Between May 1999 and June 2001, a total of 437 vessels including 364 yachts, 38 commercial fishing trawlers and 35 apprehended illegal vessels have been inspected by APM. The 35 apprehended vessels were identified as a high-risk category of vessels following the finding of significant black striped mussel fouling on the hulls of apprehended vessels moored within Darwin Harbour at a quarantine area during an APM monthly survey of high traffic areas in Darwin Harbour. Apprehended vessels typically travel from Southeast Asia, which is renowned for its biological diversity, including several undesirable marine species such as the black striped mussel and Asian green mussel. New protocols have been developed for apprehended vessels, and the Australian vessels that come into contact with them, to reduce the chance of them being moored close onshore before inspection. A plan to scuttle fouled apprehended vessels at sea, in areas where the risk of larvae reaching the coast are minimal, is under consideration in light of legal obligations.

The vessel inspection programme has intercepted at least four undesirable taxa: a variety of bryozoans (not identified to species), and three molluscs: *Musculista senhousia*, *Perna viridis* and *Mytilopsis* sp. However, monthly photographs of designated underwater surfaces and concurrent monthly checking of settlement collectors for the appearance of exotic fouling organisms have determined that the high traffic areas of Darwin Harbour, its marinas and Gove Harbour remain free of alien marine species that are known to be invasive.

ACKNOWLEDGMENTS

Eradicating *Mytilopsis* sp. from Darwin was a whole of government approach, led by the Northern Territory Government. J. Daulby (Assistant Commissioner for Police, Fire and Emergency Services) coordinated and led the response team. R. Smith (Chief Executive Officer, Dept. Primary Industry and Fisheries) developed the overall management and treatment plans. N. Rayns (Director of Fisheries) led the operational response to the *Mytilopsis* sp. in the marinas. R. Pyne (Deputy Director Fisheries) assisted by B. Russell and R. Willan (Museum and Art Gallery Northern Territory) led the diver surveys. C. Shelley (Manager Aquaculture) led, and N. Munksgaard and A. Marianelli (Northern Territory University) assisted development of the chemical treatments and physiochemical monitoring of the marinas. J. Munday (Director Media Relations and Corporate Communications, NT Police, Fire and Emergency Services) led the media and public communications response. Identification material and kits were rapidly provided by R. Martin and S. Spinks (CSIRO Marine Research). I. Kilduff and G. Tucker (Australian Quarantine and Inspection Service) and G. Mayer and S. Srinivas (Marine Branch, Department of Transport and Works) led the vessel tracking and operational aspects of vessel treatment. M. Campbell, C. Hewitt and R. Gurney (CSIRO Marine Research) planned and conducted the monitoring of the treatments in the marinas using caged mussels. G. Rawlin (Agriculture, Fisheries and Forestry Australia) coordinated the national response. In addition to those listed above, over 260 other persons achieved what at several points appeared to be the unachievable.

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The eradication of alien mammals from five offshore islands, Mauritius, Indian Ocean

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Abstract Following the removal of rabbits from Round Island in 1979 and the publication of a management plan in 1989, the Mauritius Government contracted Wildlife Management International Limited in 1993 to fulfil one of the plan's recommendations to survey the offshore islands of Mauritius and Rodrigues and to prepare an offshore islands management plan. This plan made a number of recommendations and priorities in relation to the removal of alien species. In 1995 work on the priorities began with the removal of Norway rats (*Rattus norvegicus*) and hares (*Lepus nigricollis*) from Gunner's Quoin, ship rats (*R. rattus*) from Gabriel Island and mice (*Mus musculus*) from Ile Cocos and Ile aux Sables. In 1998 cats (*Felis catus*), ship rats and mice were removed from Flat Island and rabbits (*Oryctolagus* sp.), which had been illegally released following the earlier eradications, from Gunner's Quoin. These programmes were hand-laid operations. In all cases the main bait was grain-based pellets containing 0.02gm/kg brodifacoum. The bait was set out on at least half of the maximum grid recommended for the rodent species targeted. The exception was cats, which were trapped in leg-hold traps. Plans are being considered for the re-introduction of reptiles and birds. Some planting of native trees has begun. This paper covers the eradication sector of the management plan.

Keywords Ship rat; *Rattus rattus*; Norway rat; *Rattus norvegicus*; mouse, *Mus musculus*; black-naped hare; *Lepus nigricollis*; feral cat; *Felis catus*; brodifacoum.

INTRODUCTION

The islands of Mauritius have been heavily modified by man and there have been a number of extinctions (Cheke 1987). Conservation work began with efforts to save some highly endangered species, notably the Mauritius kestrel, *Falco punctatus* (Jones *et al.* 1994) and pink pigeon, *Columba mayeri* (Jones and Hartley 1995). It was recognised early that the preservation of what remained of natural habitats was extremely important if the long-term viability of many species was to be assured. This began in two distinct areas, on the offshore islands and in the highlands indigenous forest that still remained. In this paper we consider the first of these habitats – the islands.

Efforts began primarily on Round Island and Ile aux Aigrettes, the first to save the last remnant of the palm savannah of northern lowland Mauritius and the latter, the lowland hardwood forest. Both habitats were in very poor condition.

Round Island was affected by the presence of two browsing species – goats (*Capra hircus*) and rabbits. The goats were removed finally in 1979 by shooting. An unsuccessful rabbit eradication attempt involving shooting was mounted in 1976 (Bullock 1977). The removal of rabbits was delayed for several years due to an organised objection by the Universities Federation for Animal Rights. They pressured the British Government who was to fund the eradication. They objected to the use of the poison strychnine and the funds were withheld.

In 1982 the Jersey Wildlife Preservation Trust raised the possibility of rabbit removal with the New Zealand Wildlife Service. The feasibility study was undertaken in 1984 (Merton 1985) and in 1986 the eradication was carried out (Merton 1987). A management plan for the island was prepared in 1989 (Merton *et al.* 1989). This included a recommendation that the other offshore islands of Mauritius be surveyed with a view to protecting their natural values.

Wildlife Management International Ltd (WMIL) was contracted by the Mauritius Government with funding being

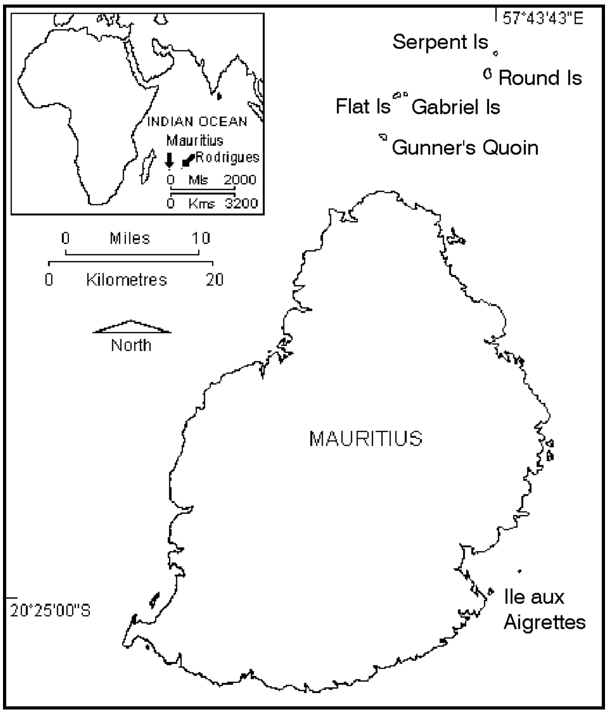

Fig. 1 Location of islands around Mauritius.

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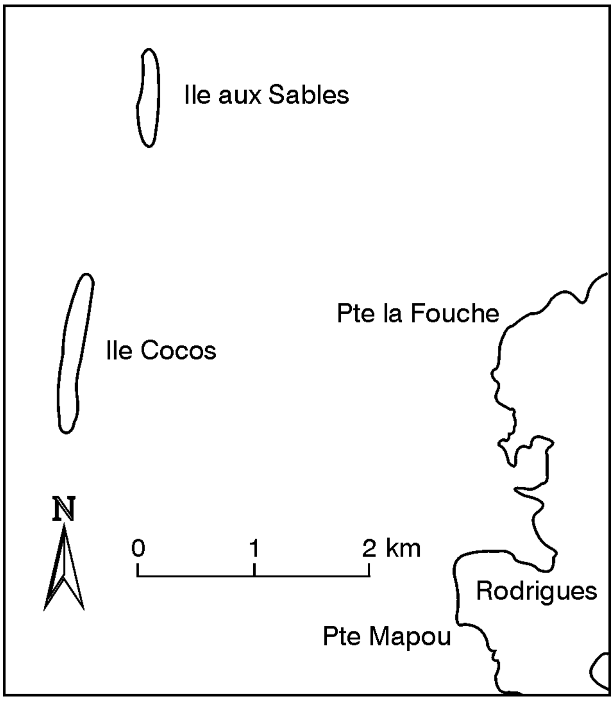

Fig. 2 Location of Ile Cocos and Ile aux Sables adjacent to Rodrigues Island.

provided by Overseas Development Agency, (now Department for International Development) United Kingdom. All of the islands off Mauritius and Rodrigues were visited. In addition to recording the flora and fauna, recommendations were made for both the long and short-term management. Most of the short-term management involved the removal of introduced mammals (Bell *et al*. 1994).

The more valuable islands, both for their biological values and for public use, were given priority. These included Gunner's Quoin, Flat and Gabriel Islands to the north of Mauritius (Fig. 1), Ile aux Aigrettes (Fig. 1), Ile aux Fouquets and Ile de la Passe to the south-east and on Rodrigues, Ile Cocos, Ile aux Sables (Fig. 2) and Crab. Some finance was available to action the recommendations, but this was very limited.

METHODS

The eradication programmes occurred in 1995 and 1998. Four islands, Gunner's Quoin, Gabriel, Ile Cocos and Ile aux Sables, were treated in 1995 and two islands, Flat and Gunner's Quoin, were involved in 1998. Each is covered separately below. Table 1 lists the islands involved and the details of the eradications.

1995 Programme

In 1995 WMIL sent a team to carry out eradications on Gunner's Quoin, Ile Cocos and Ile aux Sables and to trial bait for Indian house shrews (*Suncus murinus*) on Ile aux Aigrettes. These had been selected as the highest priorities during the island survey project.

Gunner's Quoin

The eradication programme began on Gunner's Quoin (65 ha). Much of the island was covered in low thorn scrub and it was necessary to cut grid lines. This was done on a 25 m grid even though we were removing Norway rat; the usual grid size for eradicating Norway rats is 100 m. The smaller grid enabled the rats to gain access to the bait more quickly and reduced the overall time of the programme. The cutting party consisted of five WMIL staff, two Mauritius Wildlife Foundation (MWF) volunteers and two Government of Mauritius National Parks Conservation Service (NPCS) staff plus a contingent from the Special Mobile Force (Mauritian Army). Bait was laid as the grid lines were cut over a block. The main bait used was Pestoff Rodent Bait 20R (a grain-based pellet made to the Wanganui No. 7 formula containing 0.02g/kg of brodifacoum and dyed green). This was supplemented with Rentokil Rid Rat (a wax block with grain containing 0.05g/kg bromadiolone and dyed green) as a back up. The sowing rate was approximately 15 kg per hectare.

There were no formal bait stations as such, the bait was just laid on the ground and the position marked with a numbered plastic tag. Laying directly on the ground is practical if the weather and soil are dry enough so that the bait maintains its shape and attractiveness. It is also only practical when there are no important non-target species.

The black-naped hare (*Lepus nigricollis*) was not originally targeted, as we did not expect them to take the bait. Once it became obvious that hares were taking the bait, this was laid wherever we found hare sign.

Ile Cocos and Ile aux Sables

Cocos and Sables Islands are two small coral sand islets (15 and 8 ha respectively), which were infested with mice. The vegetation was primarily grassland (*Stenotaphrum dimidiatum*) with patches of *Pisonia grandis* and *Casuarina equisetifolia* trees. Ile Cocos is a tourist destination and has high numbers of breeding noddies, *Anous stolidus* and *A. tenuirostris*. There is a warden station on Ile Cocos.

Here we used a 10 m grid and 25 mm plastic tubes as bait stations where land crabs were dense. The same bait as on Gunner's Quoin was used.

Ile aux Aigrettes

To determine whether it was possible to eradicate Indian house shrew from Ile aux Aigrettes some trials were undertaken using newly-developed bait. A fish meal and vegetable oil paste with 0.05 mg brodifacoum per kg were laid at 14 sites around the island. The bait stations were laid in the late afternoon and left overnight. The stations were observed during the evening and any shrew activity was noted. The stations were removed the following morning and checked for shrew sign.

Table 1 Eradication timing, method, target species and island involved during the eradication programmes in Mauritius.

ISLAND	SIZE (ha)	TARGET SPECIES	METHOD	TIMING
Gunner's Quoin	65	Norway rat *Rattus norvegicus*	Hand-laid 25 m grid Pestoff 20R & Rid Rat. 15 kg/ha	26/9 - 26/10/95
Gunner's Quoin	65	Black-naped hare *Lepus nigricollis*	Hand-laid. Isolated bait stations (areas of hare activity) Pestoff 20R. 15 kg/ha	26/9 - 26/10/95
Gunner's Quoin	65	Domestic rabbit *Oryctolagus* sp.	Hand-laid. Isolated bait stations (areas of rabbit activity) Pestoff 20R. 10 kg/ha	Undertaken by NPCS in late 1998
Ile Cocos	15	Mouse *Mus musculus*	Hand-laid 10 m grid Pestoff 20R. 10 kg/ha	1/11 - 21/11/95
Ile aux Sables	8	Mouse *Mus musculus*	Hand-laid 10 m grid Pestoff 20R. 10 kg/ha	1/11 - 21/11/95
Gabriel Island	42	Ship rat *Rattus rattus*	Hand-broadcast Pestoff 20R & Rid Rat. 20 kg/ha	27/11 - 29/11/95
Flat Island	253	Ship rat *Rattus rattus*	Hand-laid 25 m grid Pestoff 20R 15 kg/ha	1/9 - 28/10/98
Flat Island	253	Mouse *Mus musculus*	Hand-laid 25 m grid Pestoff 20R 15 kg/ha	1/9 - 28/10/98
Flat Island	253	Feral cat *Felis catus*	Leg-hold traps Secondary poisoning	1/9 - 28/10/98

Gabriel Island

Before departing from Mauritius we decided to lay left over bait on Gabriel Island (42 ha) in an effort to remove the very high incidence of ship rat. Gabriel Island is covered mainly with a low endemic shrub (*Psiadia arguta*) with some *Lantana camara*. About 50% of the island was grassland. The bait was laid in a day using a team of 12 persons spread out in a line broadcasting the bait as they went. Both Pestoff Rodent Bait 20R and Rentokil Rid Rat were used. The sowing rate was approximately 20 kg per hectare.

As the island is very close to Flat Island it was recommended that eradication be carried out there within 12 to 18 months to prevent re-colonisation of Gabriel Island.

1998 Programme

The Outer Islets Advisory Committee recognised that the recently cleared Gabriel Island was at risk from re-infestation as long as rats were still present on neighbouring Flat Island. A lagoon separates the islands by some 500 m but at low water the distance is minimal because of the exposed reef. WMIL was contracted by the Government of Mauritius to eradicate all alien mammals on Flat Island – ship rat, mouse and feral cat.

Flat Island

About half of Flat Island (253 ha) is covered with trees and scrub. This meant that establishing a grid required lines to be cut before bait was laid.

The grid was set at 25 m because of the presence of mice. The grid took four weeks to establish and there were 3500 bait stations. The party consisted of six WMIL, two NPCS and two MWF staff. The bait was laid directly on the ground despite the high level of crabs (*Coenobita* sp. and *Cardisoma* sp.), apart from nine stations where bait was placed in round tubes raised off the ground.

The only bait used was Pestoff Rodent Bait 20R. This was laid in five pulses, with a break of two to three days between each lay.

Cats were only targeted toward the end of the programme as we expected a number to succumb to secondary poisoning. Leg-hold traps were used and were placed where a cat had been sighted or fresh cat sign was present.

Gunner's Quoin

Before we left New Zealand for the Flat Island project we were advised that although earlier reports had indicated hares had been removed from Gunner's Quoin, this now appeared to be an error as heavy browsing and sign had been seen on one of the monitoring trips. Extra bait was taken so that another attempt to remove the hares could be made. An inspection prior to starting Flat Island revealed that the culprits were not hares but rabbits which must have been released subsequent to the removal of hares. Some of the hunting fraternity was suspected as they used to visit the island to shoot hares. WMIL did not have time after completion of Flat Island to lay the bait but advised NPCS who undertook to spread the bait.

Problems

The work was without any serious problems. The only complication was the high numbers of crabs. On parts of Ile Cocos and Ile aux Sables high concentrations of large land crabs occurred where there were low-lying embayments, which flood in extreme high tides. In such sites we set out bait in plastic tubes as described earlier. This did not deter the crabs, which took both the bait station, complete with bait, down into their holes. After two days the empty bait station would be pushed out of the burrow. In addition to using bait stations, some bait was broadcast in these areas to ensure there was some bait still available to mice if they were present.

Crabs (both land and hermit) were also present in large numbers on Gabriel Island. It was thought these might cause a possible failure of the eradication. To overcome this we laid a heavier than usual amounts of bait (20 kg/ha).

On Flat Island both crabs were present in a section of the island but it was found necessary to raise only nine bait stations about 15 cm above the ground to exclude them. However earlier experience showed that the best way to overcome crabs (at least where there are no non-target species at risk) was to increase the amount of bait applied.

RESULTS

Post eradication monitoring has been carried out mainly by NPCS. They have regularly reported on the success of the eradications and are discussing options for the future use of the islands, many of which had been suggested in the report on the Survey and Management Plan for the Outer Islands (Bell *et al.* 1994). Some recovery of the vegetation had been noted even before we left the islands.

Gunner's Quoin

During the eradication, rats began to die within two days and bait take had ceased within a week, apart from areas where hares were taking bait. A few dead hares were found but at least one hare was still alive when we left the island. Bait was still available to any survivors. A check before we departed from Mauritius showed no evidence of any hares present. This was confirmed later during visits by NPCS.

The return visit and bait lay for rabbits in 1998 resulted in their removal. It appeared that these animals were the domestic strain as most were coloured either black, black and white or fawn. It is a good example of the need to ensure that no immigration, either accidental or deliberate, occurs after eradication.

The first changes were noted before the eradication team left the island. The vegetation was responding to the lack of browsing but the most interesting observation was the increased visibility of the skinks (*Scelotes bojerii* and *Cryptoblepharus boutonii*). These lizards seemed to have responded to the removal of their main predator, the rat, and spent more time in the open rather than in rock crevices.

The NPCS monitoring trips recorded a strong seedling growth of *Dracaena concinna*, *Latania loddigesii* and *Pandanus vandermeerschii*. Previously rats destroyed most, if not all, the seeds. The other major find was the rediscovery of the night gecko (*Nactus coindemirensis*) which was only known from two previous specimens. It is now regularly seen.

The revegetation was heavily browsed as a result of the introduction of rabbits but is now improving again. Along with the indigenous plants, the weed species were also flourishing and some, such as a creeper (*Cissus* sp.), may become a problem. A systematic control programme may be necessary.

Cocos and Sables Islands

The only area where mice were noticeable was at the warden station. These soon disappeared as the eradication programme continued. To date there have been no spectacular changes on Cocos and Sables Islands.

Gabriel Island

The WMIL team had departed Mauritius before this eradication could be confirmed. Later checks by NPCS proved the eradication had been successful. The most noticeable change on Gabriel Island has been the regeneration of *Pandanus* and *Latania*, which had not occurred for many years. There also appears to have been an increase in wedge-tailed shearwaters (*Puffinus pacificus*) and red-tailed tropicbird (*Phaethon rubricauda*) but there has been no formal assessment.

Flat Island

During the eradication bait take was heavy after the first baiting (80-90%). On the second and third pulses, only about 10% of stations had been visited and after the fourth, bait was taken only from less than 10 stations. The final baiting was untouched apart from that taken by insects, crabs and lizards. One cat was seen and this animal was successfully trapped the same night, using a leg hold trap and was destroyed. Only one other cat was found, dead, and this had almost certainly been a victim of secondary poisoning. It is probable that the cat population was very small although sign showed they were roaming the whole island. They most likely originated from animals left behind either deliberately, to keep rats down around camps (camp areas used by day trippers) or accidentally, when pets were brought on to the island and they could not be found before leaving.

Ile aux Aigrettes

The trial baiting for shrews on Ile aux Aigrettes was not successful. It was thought that they could detect the 'bitrex' – an additive used in some brodifacoum-based poisons and this put them off taking the bait. Brodifacoum had been proved a suitable poison for shrews (Morris and Morris 1991). However bait trials they carried out showed that there had to be a very high consumption of bait if the shrews were to ingest a fatal dose at 50 ppm. This may mean that to be effective in the field, higher concentrations are needed and chemical companies have expressed reluctance to supply these doses.

More trials to find attractive bait should be undertaken before any further attempt is made to poison the shrews.

CONCLUSIONS

The removal of rats and subsequent improvement in the vegetation on Gunner's Quoin opens the opportunity to re-establish or translocate some threatened species. Because of the limited range of trees and shrubs, it is likely that some of the restricted lizards (e.g. Guenther's gecko (*Phelsuma guentheri*) and Telfair's skink (*Leiolopisma telfairii*)) could be the first choice. Later when the vegetation has recovered further (after 10yrs?) some of the threatened smaller insectivorous passerines could be introduced (e.g. Mauritius fody (*Foudia rubra*) and Paradise flycatcher (*Tersiphone bourbonensis desolata*)).

Unfortunately there have been no spectacular changes on Ile Cocos and Ile aux Sables. These islands are too small to provide habitat for some of the Rodrigues threatened species. A larger island, such as Crab, would need to be restored if new habitat was to be provided for the local endemic warbler (*Acrocephalus rodericanus*) and fody (*Foudia flavicans*). At 15 ha Ile Cocos may be a suitable site for the re-introduction of these birds as they can exist at very high densities if food supply is adequate. Intensive planting of native plants would have to occur before this can happen to ensure food availability.

As the largest offshore island, Flat Island has the greatest potential for use in threatened species management. The island's vegetation is primarily introduced trees and shrubs but some planting of native species has begun. Even without further planting the island could be used at present for some of the threatened species, such as Mauritius fody, flycatcher and perhaps olive white-eye (*Zosterops chloronthos*). The introduction of the Mauritius cuckoo shrike (*Coracina typica*) and bulbul (*Hypsipetes olivaceus*) could follow later as the forest cover improves and is made more diverse.

Flat Island is a popular tourist and picnic site. This aspect has to be taken into account when planning the continuous rodent-free state of the island and future translocations and monitoring. The development of a management plan for this island and Gunner's Quoin will address these aspects. This is scheduled for the near future.

The Mauritius Government and its NGO collaborator, MWF, are to be commended for their far-sighted vision to not only remove alien mammals from their important islands, but also for establishing formal management plans and following these through with active management.

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The eradication of possums from Kapiti Island, New Zealand

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Abstract The Australian marsupial the brushtail possum *Trichosurus vulpecula* was introduced to Kapiti Island, a nature reserve off the south-west coast of New Zealand, in 1893. Various attempts were made to control possums on Kapiti because of their negative impacts on forest ecosystems. Possums can kill individual trees, potentially alter forest succession and regeneration processes, suppress flowering and fruiting, and prey upon native birds and other native animals. Possum control was initiated in 1980 and approximately 21,000 possums were removed by 1985. Eradication was achieved using traps, dogs and guns. Dogs located 32 of the 80 possums that were removed during 1985-1986. This programme has shown that dogs and intensive trapping are effective tools for eradicating possums from large areas of land where re-invasion is prevented. Eradication attempts are inherently risky and require a bold commitment from those bureaucracies with the responsibility to succeed. Thorough planning and highly skilled and motivated teams are essential ingredients to the success of eradication attempts.

Keywords Leg-hold traps; trained dogs.

INTRODUCTION

The Australian brushtail possum (*Trichosurus vulpecula*) was introduced to New Zealand in 1840 and subsequently released at many different sites throughout the country by Acclimatisation Societies and private individuals to establish a fur industry (Pracy 1974). The possum has been a very successful colonist, reaching high densities in favoured habitats (>30 possums/ha), and occupying a wide range of habitats throughout New Zealand (Clout and Ericksen 2000). Possums are pests that have had dramatic impacts on native plant and animal communities, and ecological processes, and they impact on agricultural production through the spread of bovine tuberculosis.

Possums were controlled on Kapiti Island by trapping between 1920 and 1968 to protect conservation values. The value of this control was disputed and a moratorium was placed on trapping in 1969, when research commenced to better quantify the importance and nature of possum impacts (Cowan 1992).

Atkinson (1992) studied possum impacts on native vegetation on Kapiti Island between 1969 and 1980. He observed "increasing defoliation, and sometimes mortality, of species vulnerable to possums" and he concluded, "had this continued, major changes in the structure and composition of the island's forests would have followed". Possums compete with birds and insects for food (foliage, flowers and fruit). They are also known to prey on eggs, chicks and adult birds (Brown *et al.* 1993), and have contributed to the local extinction of North Island kokako (Innes *et al*. 1999). However, at the time that research was carried out on Kapiti Island little was known of their impacts on birds. Phil Cowan was supported by a range of other scientists and Lands and Survey staff when he proposed eradication of possums following the success of possum control between 1980 and 1982 (Cowan 1982).

Four government departments with overlapping responsibilities were involved in the debate over the justification and feasibility of eradication. The eradication of cats from Little Barrier Island in 1980 (Veitch 2001) offered psychological weight in favour of eradication (B. Bell pers. comm.). The New Zealand Forest Service was the one department that favoured sustained control over eradication but still contributed significantly to the eradication programme, primarily due to political pressure. This debate was not fully resolved at the time but a commercial operation turned into intensive control that eventually evolved into an eradication programme (at least in the minds and hearts of those on the ground). Possums were eradicated from Kapiti Island in October 1986.

The eradication of possums from Kapiti Island has been described elsewhere by Cowan (1992) in a scientific paper and some of the methods by Sherley (1992) in a published report. This paper differs from Cowan (1992) by describing more fully the methods used and from Sherley (1992) by expanding on the lessons learned. We describe in detail the methods used and their relative importance, and outline the key ingredients for success that can be applied elsewhere to eradicate possums and other introduced pests.

KAPITI ISLAND

Kapiti is a rugged island lying 5 km off the south-west coast of the North Island of New Zealand. It has precipitous western cliffs that run along a fault line and numerous streams and gullies dissect its eastern slopes. Tall podocarp forest once covered Kapiti but it was largely deforested by Maori and European fires and farming in the 19th and

Pages 46-52 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

early 20th century (Maclean 1999). The island's vegetation has regenerated (assisted by plantings and introductions by caretakers) in the 20th century and the island is now a mosaic of forest and shrubland with some grassland. Tawa (*Beilschmiedia tawa*) and kohekohe (*Dysoxylum spectabile*) are the dominant canopy species but shrublands dominated by kanuka (*Kunzea ericoides*) and fivefinger (*Pseudopanax arboreus*) are the most common vegetation type (Atkinson 1992). Kapiti was gazetted a nature reserve in 1897.

Kapiti is one of New Zealand's most exciting restoration stories. At 1965 ha, it is New Zealand's second largest offshore nature reserve and the largest single area of lowland coastal forest that is free from introduced mammalian herbivores and predators. It provides a home for various endangered bird species including little spotted kiwi (*Apteryx owenii*), saddleback (*Philesturnus carunculatus*), kokako (*Callaeas cinerea*), hihi (*Notiomystis cincta*) and takahe (*Porphyrio mantelli*). Cattle (*Bos taurus*), sheep (*Ovis aries*), goats (*Capra hircus*), pigs (*Sus scrofa*) and feral cats (*Felis catus*) were eradicated from Kapiti between 1916 and 1934 (Veitch and Bell 1990; Maclean 1999). Possums were introduced to Kapiti Island in 1893, just four years before Kapiti became a nature reserve, and were eradicated in 1986. Norway rats (*Rattus norvegicus*) and Pacific rats (*R. exulans*) were eradicated in 1996 (Empson & Miskelly 1999). With its prolific birdlife and healthy forest, Kapiti Island offers up to 18,000 visitors a year a window into how New Zealand used to be.

METHODS

Three phase operation

There were three distinct phases to the eradication of possums from Kapiti Island:

- Phase 1. Feb 1980-Oct 1982. Seven trappers commercially harvesting possum skins. Peter Daniel, the Kapiti Island ranger, supervised trappers to minimise the risk to birds and strongly discouraged normal harvesting practices such as release of small possums. Tracks were cut by the trappers along major ridges, spurs and valleys, mainly in the centre of the island.
- Phase 2. Feb 1983-Jan 1985. Four trappers worked for wages as opposed to selling skins and were assisted by Wildlife Service trainees. The complex network of tracks was completed during this phase and trapping intensified from 800 to 1500 traps set each night.
- Phase 3. Mar 1985-Mar 1987. Two trappers working up to 1800 traps, and three dog handlers with teams of up to three dogs intensively searched the island for the remaining possums.

Tracks, traps, sets and lures

An extensive network of tracks totalling more than 450 km covered Kapiti by the end of the intensive trapping phase. Tracks were cut 50-80 metres apart so that all possums had access to traps placed along tracks within their home ranges (Cowan 1992). Tracks on the western cliffs were the exception at up to 400 m apart because of the difficulty of the terrain. Tracks were cut so as to create minimal disturbance to the native vegetation but allow a trapper with sets protruding above their heads to have unimpeded access.

Lanes Ace leg-hold traps were set at approximately 50 m intervals along all tracks. Traps were checked daily and regularly maintained to minimise the risk of possums escaping and potentially becoming trap shy. Traps were sprung approximately once a week and the mechanism checked and CRC lubricant applied when required. Most importantly, traps were left in place for up to six months. It was believed that individual animals that were aware of the traps (potentially trap shy possums) would eventually make a mistake. Traps were not "fine set", instead they were set firm enough to minimise the risk of small birds (e.g. robins (*Petroica australis*)) being caught.

Traps were moved from the south to the north end of the island in a "rolling front" (i.e. the southern-most traps were placed in front of the northern-most traps). Trappers took approximately two years to traverse the island and did so twice between February 1983 and December 1986. Karaka (*Corynocarpus laevigatus*) groves were also trapped each February to target possums that would travel long distances to feed on ripe fruit.

Wooden sets (Fig. 1) were used to minimise the risk to flightless birds (little spotted kiwi and weka (*Gallirallus australis*)) by raising traps out of their reach. The sets were designed to be effective at catching possums and this set design evolved through time. The final design provided a cradle in which traps were inset flush with the lower part of the set and offset so that the paws of the possum used to cling to the sides of the set when climbing would be in line with the trap. Traps were firmly held to minimise the risk of being knocked off by possums but traps still jumped when sprung, which increased efficacy.

Traps were tied to staples in trees with a self-tightening knot (to prevent possums escaping with traps) and at a height that allowed possums to sit on the ground to minimise their distress and desire to escape. Sets were placed on palatable tree species with the base of the set on and in line with a possum run to increase the likelihood of use. Concentrated liquid cinnamon essence was the most commonly used lure because it was believed to be effective and attractive to possums but not to birds (Sherley 1992). Subsequent trials have confirmed that cinnamon is an effective and safe lure (Morgan *et al.* 1995). Cinnamon essence was placed on the tree above the top of the set (every five days and after heavy rain) so that a possum smelling it would place its paws on or near the trap.

Huts, helicopters and boats

Trappers and dog handlers used a combination of portable and permanent huts to facilitate quick access to all parts of the island. Multiple camps saved time and money and enabled the staff to achieve more. This was particularly important to dog handlers working at night. Helicopters and boats were essential tools for the movement of

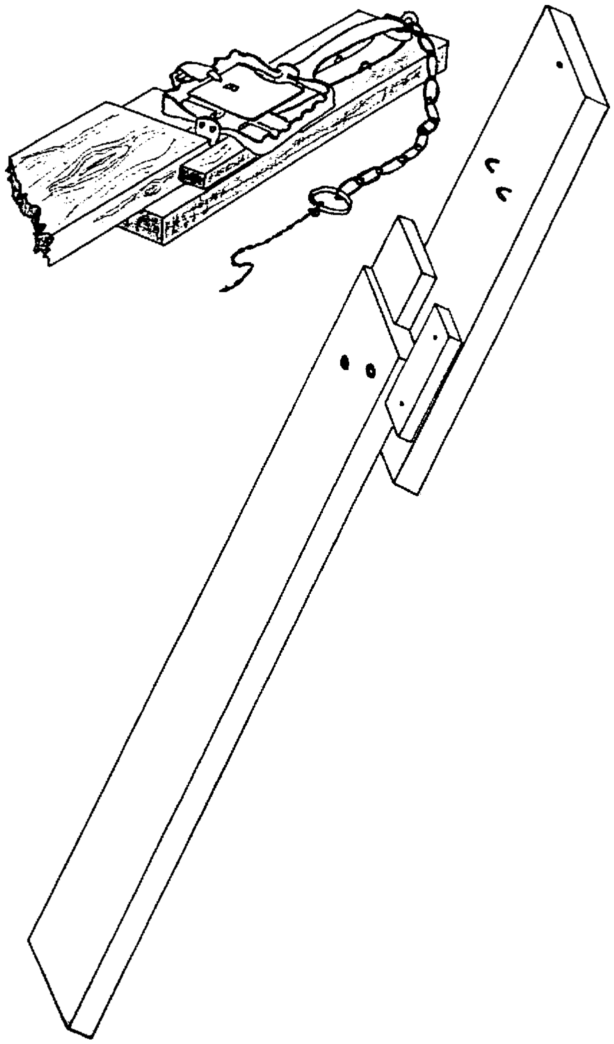

Fig. 1 Kapiti Island possum trap set, designed by Geoff Alexandra (from Sherley 1992).

huts, equipment and personnel. The extensive track system also greatly reduced travel time.

Dogs, handlers and guns

Dogs were used in the eradication phase only (Phase 3). The island was divided into blocks of approximately 40 ha and these blocks were worked for about 10 days and three fine nights. Initially all tracks were walked and then the ground between tracks was walked in an intensive grid pattern. Dogs and handlers worked closely, in constant contact, as a team. The dogs were trained to respond to possum scent, and to ignore the scent of non-target species such as birds and rats. This skill was achieved by teaching the dogs to associate positive behaviour with the word 'yes' (e.g. eating food and killing possums) and negative behaviour with the word 'no'. Kiwi, weka and rats were placed in front of the dogs and the word 'no' was repeated in harsh tones. Non-target training was reinforced regularly in the field when non-targets were encountered and by the exposure of dogs to non-targets captured for that purpose. Dogs and handlers left the island periodically to reinforce dog/handler possum hunting skills on the Kapiti coast and in the Wairarapa. This reinforcement regime was especially important when possum encounters on Kapiti Island were extremely low, as was the case in the late stages of the eradication phase.

The dogs walked quietly ahead of handlers but within view. Dog behaviour changed dramatically once possum scent was detected. As the dog actively followed the scent it was the dog handler's job to stay with it and call the dog back if required. Once the possum was located the dog and handler worked as a team to place the possum at maximum risk from gun or dog (e.g. the dog could be directed to where the possum was seen to break out of cover or move through vegetation above ground). Guns (mostly shotguns) were used to kill possums up trees and the dogs were trained to kill possums they caught on the ground.

Three dog handlers generally worked with teams of two dogs each in the field (though each handler had up to four dogs available). Dogs were of mixed breeds but small terriers were favoured because of their tenacity and ability to move quickly through tight scrub. Handlers swapped blocks on completion so that each handler checked each block and therefore the whole island. Dogs and handlers generally worked behind the rolling front of traps. They were allowed to search for possums ahead of their blocks when scent led from the block. The dogs caught most possums in areas that had not previously been systematically hunted with dogs while travelling north for dinner at the ranger's house or to leave the island. No possum was caught in a block that had previously been worked by another handler. Night hunting was by torchlight along tracks until scent was located and then it was a matter of the hunter keeping up with the dogs. Not all dogs 'trail barked' (barked as they ran following fresh possum scent) but trail barking dogs were an advantage. A spotlight was used to locate the possum once it was forced to hide in a tree by the pursuing dog(s).

Following the first complete sweep of the island by dog hunting teams, all three handlers walked all tracks on the island with their dogs and no possums were scented. One trained dog remained after the eradication and was used to repeatedly search the island while track markers and sets were removed.

Aerial and bait station application of 1080

Approximately 330 ha at the northern end of the western cliffs were sown by helicopter in August 1984 with 15 kg/ha of sieved carrot baits that were dyed green and impregnated with 0.15% 1080 (mono sodium fluoroacetate) poison. Because Global Positioning Systems were not available in 1984, visual markers were used to guide the application of baits. One pre-feed of non-lured carrot baits (dyed green) was spread on the cliffs by helicopter one week before the poison drop.

Originally it was planned to follow phase two of trapping with poisoning. Bird-proof poison bait stations containing 1080 paste were tested for approximately six months during phase two. However poisoning was abandoned because so few baits were taken relative to the effort involved (Sherley 1992). With hindsight, the low bait take is not surprising given that possums were then at such low density (1 per 25 ha).

Result monitoring

Mark-recapture of possums was used during all phases of the control and eradication to estimate the success of various phases of the operation (Cowan 1992). Two cage trap lines were used that were previously established and run by DSIR during 1975-1980. Faecal pellet analysis and trap catch data were also used to estimate the numbers of possums remaining in the eradication phase. These methods were not highly sensitive (far more sampling effort would have been required to give more accurate measures of abundance) but gave approximate measures of possum abundance. Also, non-target kills were monitored throughout the control and eradication operation (Cowan 1992).

Dogs provided an excellent tool for detecting possums at low density and they had the added advantage of being able to determine when no possums remained.

RESULTS

Gin trapping was a very effective tool for possum removal when set at the density and for the duration of time used on Kapiti (Cowan 1992). Gin traps removed over 19,500 possums over the duration of this programme (Table 1). In addition, based on pellet counts, an estimated 1500 possums were killed in the aerial 1080 operation at the northern end of the western cliffs. A further 32 possums were caught by dogs. Many possums caught by the dogs were old and showed signs of having escaped from a trap and were probably trap shy (K. Brown pers. obs.).

Non-targets

A total of 181 birds were caught in traps (Cowan 1992). Not all birds were killed and many were released with minor injuries. The most common species caught were New Zealand pigeon (*Hemiphaga novaeseelandiae*) (70), morepork (*Ninox novaeseelandiae*) (47), weka (29) and kaka (*Nestor meridionalis*) (16). An unknown number of birds were killed in the aerial 1080 operation but searchers found only three.

Table 1 Possums caught and trapping effort on Kapiti Island between February 1980 and October 1986 (adapted from Cowan 1992).

Date and phase	Number traps set	Number possums caught	Percent trap success
1. Feb 1980-Oct 1982 (Commercial trappers)	65,866	15,631	23.7
2. Feb 1983-Jan 1985 (Intensive control)	589,336	3933	0.667
3. Mar 1985-Dec 1986 (Eradication)	743,538	48	0.007
Total	1,398,740	19,612	0.014

DISCUSSION

Three basic criteria have been identified that must be met if eradication is to be successful; the rate of removal exceeds the rate of increase, there is no immigration, and all animals must be at risk (Cowan 1992; Bomford and O'Brian 1995; Parkes 1996). However there are many other strategic and operational factors that will determine the feasibility and efficiency of any particular eradication attempt and whether the above criteria can be met. Strategic and operational factors that are important to the success of the Kapiti operation are relevant to other eradication attempts and are discussed below.

Is eradication a viable option?

Pest managers need to decide between eradication, control and doing nothing to effectively manage a pest species (Bomford & O'Brian 1995; Parkes 1996). The decision to eradicate a pest will be dependent on people's assessment of the feasibility and justification of eradication. Lands and Survey (which administered the island), Department of Industrial and Scientific Research – Ecology Division and Botany Divisions (which researched Kapiti possums and their impacts), and Internal Affairs – Wildlife Service (protected species management) were the key agencies involved that believed possum eradication was feasible and justifiable. Their arguments were that the island flora and fauna were of national significance; that possums competed for food with, and preyed on, native birds, and that forest structure and individual species of plants were under threat. They also believed that eradication was feasible and was far more cost-effective (in the long-term) than sustained control. Cats were eradicated from Little Barrier Island (2817 ha) in 1980 (Veitch 2001) and this provided psychological support for the attempted eradication of possums from Kapiti Island (B. Bell pers. comm.).

The New Zealand Forest Service (responsible for wild animal control) argued that local possum eradication had never been achieved on the mainland and therefore was not feasible on Kapiti Island. The western cliffs were considered to be too steep to allow hunters access to all individual possums. The Forest Service also did not believe that the scientific evidence on possum impacts on ecosystem health was sufficient to justify eradication and it therefore supported sustained control as the most viable option. They also argued that the opportunity cost was high (i.e. valuable resources would be taken from other wild animal control operations elsewhere). Both points of view were reasonable but the lack of consensus did impact on the Kapiti possum eradication operation by fuelling interdepartmental frictions and slowing logistical support.

Iwi (indigenous people) own 12 hectares at Waiorua Bay at the northern end of Kapiti Island. These owners allowed access to their land and provided support and encouragement for the eradication effort. Hence the first lesson from this project is that *"buy-in of all key stakeholders is very desirable if not essential"*.

Institutional involvement and logistic support

Despite the Forest Service resistance to eradication, the inter-departmental working party fought hard to ensure ongoing funding. Logistical support was provided by the Forest Service based in Masterton and Lands and Survey through Peter Daniel, the Ranger on Kapiti. Peter was tireless in his efforts to ensure resources were available when required. Peter and his wife Linda also provided much psychological support over evening meals, once a week. The DSIR provided scientific advice, evaluation processes through monitoring and moral support. The Wildlife Service provided trainees as labour early in the operation, and Lands and Survey provided one staff, and Forest Service a further four staff. Hence the second lesson from this project is that *"an ongoing commitment of staff and resources is essential"*.

No immigration and access to all animals

Possums are poor swimmers and Kapiti is five kilometres from the mainland so immigration was not feasible short of possums being carried to the island as a malicious act. Only permitted boats are allowed to land on Kapiti. Anticoagulant poison is laid on boats that carry visitors to the island and visitors' luggage is searched prior to departure from the mainland to Kapiti. These precautions are primarily designed to prevent rat invasion but also reduce the risk of the accidental transport of possums to Kapiti. Hence the third lesson from this project is to *"ensure further immigration does not occur through stringent quarantine procedures"*.

The extensive network of tracks and placement of traps every 50 – 80 metres meant that all animals had access to traps. Tracking of the western cliffs was initially thought to be impossible but was achieved and was supported by aerial poisoning. The spacing of tracks and traps was based on research of possum home range sizes (Cowan 1992). Trapping at seasonal food supplies (such as karaka fruiting) further increased the risk to possums. Intensive searching with dogs also ensured that all animals were accessible. The use of multiple camps was also important in allowing traps to be serviced and dogs to be worked at the intensity and frequency required. Regular discussions were held among trappers and progress was reviewed with possum experts. Hence the fourth lesson from this project is to *"know your target animal and place all individuals at risk"*.

The right tools to do the job

An initial "knockdown" followed by "mop-up" is a strategically sound approach to eradication. The aerial application of 1080 is a potentially useful knockdown tool but was unacceptable on Kapiti due to the perceived risk to rare and endangered bird species. Traps placed on sets off the ground were the chosen tools for knockdown and they proved to be very effective. The efficacy of trapping was placed at risk by experimentation with set designs (and trap types) that resulted in a high proportion of escapes in the early design stage. Some escaped possums became trap shy that increased the risk of eradication failure. A very effective set was eventually designed (Fig. 1) and traps were maintained to a very high standard. Hence the fifth lesson from this project is that *"eradication tools should be developed off-site to minimise the risk that target species will become shy and avoid the eradication tools"*.

Highly-trained dogs (to minimise non-target risks and maximise search and destroy capabilities) proved ideal for locating the few remaining possums that had avoided traps. The dogs killed many possums in areas of high possum density on the mainland while training but it is not known if dogs would have been more successful than traps at high possum density, early in the operation on Kapiti. Part of the dogs' success as "mop-up tools" lay in the fact that they were tracking individual possums. This enabled information to be gathered on an individual (over a two-week period in one case) that inevitably ended in it being located and killed. The dogs provided the ultimate monitoring tool because not only could they detect the presence of possums at low density, they could also confirm the absence of possums. Hence the sixth lesson from this project is that *"well tested tools should be used in the right sequence to achieve the knockdown and subsequent mop-up phases of the operation"*. The seventh lesson from this project is that *"the tools chosen should minimise non-target risks"*. The eighth lesson from this project is that *"monitoring allows progress to be tracked and provides valuable information to sustain support"*.

Team attitude

The core team on Kapiti was small, highly motivated, highly committed, physically and mentally fit, skilled, compatible and hard working. The work was physically and mentally demanding and involved carrying heavy loads (100 gin traps equals approximately 50 kg) up and down steep slippery terrain, long periods of repetitive work with few returns (especially when possums were scarce) and living and working in close confinement. Key skills included the design and building of sets, building of huts, dog training to a high standard, meticulous care of equipment, organisation of logistics and keeping accurate records. The team often worked 12-14 hour days and dog handlers regularly worked during the day and after dark. Every team member believed that eradication was inevitable. This was particularly clear to the dog handlers who understood the effectiveness of their dogs. Hence the ninth lesson from this project is that *"team skills (personal and technical) and attitude are essential ingredients in effective eradication"*.

Eradication of possums from other islands and areas of mainland New Zealand

Possums have been eradicated from Codfish Island (1336 ha), Rangitoto (2333 ha) and Motutapu Islands (1510 ha), and Allports Island (16 ha) and a number of other small islands since the eradication of possums from Kapiti Island (Clout and Ericksen 2000).

Traps, cyanide poison and dogs were used to eradicate possums from Codfish Island and a Kapiti Island dog handler removed the last possum. The eradication of possums on Codfish Island was thought to be complete in early 1987. A dog handler from Kapiti Island went to Codfish to confirm that eradication had been achieved in May 1987 for two weeks soon after possums were eradicated from Kapiti. He systematically worked the island, saving the tall forest habitat in the centre of the island for last. He detected and killed a female possum on day one of entering her territory and no other possums have been detected on Codfish Island since (R. Cairns pers. comm.).

Possums were eradicated from Rangitoto and Motutapu Islands in 1997 and 1996 respectively, using traps, dogs and guns and following the advice from Kapiti Island eradication staff (Mowbray 2002). Possums were also eradicated from Allports Island in conjunction with a mouse eradication programme (Brown 1993). Brodifacoum in Talon 50WB baits was the sole method used. Brown (1993) stated that the use of dogs would probably have been more cost-effective if possums were the only target animals (K. P. Brown had provided an estimated cost for the use of a dog to eradicate possums from Allports Island). A combination of traps and poisons was used to remove possums from most islands identified by Clout and Ericksen (2000).

It is our opinion that eradication of possums inside predator-proof fences on very large areas of the New Zealand mainland is feasible. Possums have been eradicated within a predator-proof fence protecting 250 ha of regenerating forest at Karori Reservoir using a combination of poison and trained dogs (R. Empson pers. comm.). Traps or poisons could be used as the initial knockdown tool and dogs could then be used to locate and kill the remaining individual possums and/or as monitoring tools to ensure that no possums remain. Hence the tenth lesson from this project is that *"tools and knowledge can be transferred to other eradication operations but new techniques can also prove successful"*.

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We would like to thank P. Cowan, J. Innes and C. R. Veitch for providing helpful comments on this paper. This publication reproduces much information previously presented by Cowan (1992) and Sherley (1992) with the permission of the authors. Many people made this difficult task possible. G. Alexandra, I. Atkinson, P. Cowan and P. Daniel were all instrumental in hatching, nurturing and supporting the eradication programme. G. Alexandra and R. Cairns provided the physical and psychological backbone of the operation. K. Brown, R. Cairns and M. James were the dog handlers. B. Collins, J. Oakley and G. Timlin all gave much time and effort. The inter-departmental working party was critical in maintaining bureaucratic support. At least another hundred individuals provided support in various ways. The eradication of possums from Kapiti Island was a collective act ultimately made possible by the hard work, intelligence, determination and vision of a handful of individuals.

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The impact of rabbit and goat eradication on the ecology of Round Island, Mauritius

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Abstract Round Island (169 ha) holds the only populations of several reptiles and plants that formerly occurred on Mauritius. Eradication of introduced goats (*Capra hircus*) by 1978 and rabbits (*Oryctolagus cuniculus*) by 1986 was predicted to allow increases in abundance of threatened species. Subsequent surveys have revealed the first substantial recruitment of three main tree species for over 100 years. The extent of ground vegetation increased slightly but is now dominated by non-native species and large unvegetated areas remain. Of three reptiles confined to Round Island, two showed no sustained increases but one increased dramatically. In the short term the general, predicted effects of eradication (increases in plant biomass and tree recruitment) were upheld. However unpredicted effects (differential population responses of reptiles and increasing rates of establishment and influence of non-native plants) have occurred. As a result new ecological communities are likely to develop on Round Island.

Keywords Impacts of eradication; rabbits; goats; plants; reptiles.

INTRODUCTION

Round Island (169 ha), 20 km NE of Mauritius, supports the only populations of several plants and reptiles that used to occur on the mainland. It is rodent-free, and one of the most important seabird islands in the western central Indian Ocean. Early reports indicate that a giant tortoise (*Cylindraspsis* sp.) was present. This native 'large herbivore' was replaced by rabbits (*Oryctolagus cuniculus*) and goats (*Capra hircus*) introduced in the early 19th century (Cheke 1987). The mammalian herbivores prevented tree recruitment, destroyed a hardwood forest, encouraged the open character of the vegetation and promoted the progressive ecological degradation described by many authors (e.g. Vinson 1964; North and Bullock 1986).

In terms of vegetation and non-avian species the significance of Round Island has largely been because of its relative 'naturalness' and as a refuge for threatened species. Despite the mammalian herbivores it has retained the last significant remnant of an open palm-rich forest, which supports 14 threatened plant taxa. The herpetofauna is similarly outstanding: Eight species (nine including *Cylindraspsis* sp.) are recorded, of which seven are Mascarene endemics and four are now confined to Round Island. However, within the last few decades the adverse impact of the rabbits and goats has caused the probable extinction of the snake, *Bolyeria multocarinata* last seen in 1975, reduced a hurricane palm *Dictyosperma album* var *conjugatum* endemic to Round Island to one individual, and reduced several other species to very low numbers (Bullock 1986; North *et al.* 1994).

In 1975/76 there were between 10 and 20 goats and the combination of a shooting expedition and a cyclone had temporarily reduced the rabbit population. By 1978 the few remaining goats (<5) had been shot out but by 1982 the rabbit population had recovered and the response of the vegetation was limited (North and Bullock 1986). In 1986 rabbits were eradicated (Merton 1987) and in 1989 the following short term changes were being observed or anticipated (North *et al.* 1994) increases in: ground vegetation (i.e. vegetation other than mature trees); the number and abundance of non-native plant species; regeneration and recruitment of palms and other trees; invertebrate abundance; habitat availability for, and population sizes of, arboreal *Phelsuma* geckos and the saurivorous snake *Casarea dussumieri*. After an initial pulse of tree regeneration, decreases in recruitment rate were expected as the availability of open seed bed habitats declined. Decreases were also observed or anticipated in the abundance of plants favoured by the presence of rabbits, goats or open ground. The projected loss of open ground was also expected to reduce optimum habitat availability for skink by reducing open basking areas, leading to their local decline.

After rabbit and goat eradication, and anticipated increases in two of the largest reptiles, *Casarea* and *Phelsuma guentheri*, reptile biomass was expected to increase. Reptile biomass is often exceptionally high on other seabird islands where food availability is enhanced by nutrient inputs from guano, plus carcasses and eggs. For example, on Cousin Island (Seychelles), the combined biomass of

Pages 53-63 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

three lizard species has been reported as at least 96 kg/ha, close to the maximum recorded for reptiles (Cheke 1984). In the short term, at least, there was no reason to expect fluctuations in the breeding populations of seabirds to be influenced by eradication of rabbits and goats. An increase in reptile biomass on Round Island after eradication would be expected only if habitat or food availability had been limiting when the mammalian herbivores were present.

In this paper, predicted and observed changes are compared using the results of surveys before and after the removal of rabbits and goats. Much valuable restoration management work has been conducted on Round Island since 1986, including the planting of Mauritian endemic trees. In this paper the focus is on describing unaided responses of threatened species and their habitats to eradication. The eradications are therefore regarded as an opportunity to gain understanding of ecological processes on Round Island (as suggested in Myers *et al.* 2000) and to place them in the context of the overall ecosystem (Zavaleta *et al.* 2001). Particular attention is paid to the influence of unpredicted processes in the short term (i.e. a decade after eradication of mammalian herbivores) on the long-term restoration of the island.

Scientific names and the status (non-native/native) of all plant species from 1975 to 2000 are given in Appendix 1. It is frequently difficult to separate long established introduced species from those that are native. This is especially so for pan-tropical species such as *Portulaca oleracea*, which here is considered to be native.

METHODS

Vegetation

In 1975 the most vegetated slopes of Round Island were divided into 12 study Areas (totalling *c.*102 ha) within which vegetation and reptile populations were recorded. These Areas formed the basis of comparable surveys in subsequent expeditions (1978 - partial surveys only, 1982, 1989 and 1996). Survey methods have been described previously (Bullock 1986; North and Bullock 1986). Individual *Latania loddigesii* and *Pandanus vandermeerschii* trees were assigned to one of seven and five size classes respectively where the largest (and presumably oldest) were Class 1 (see Fig. 1 and 2). Trees were counted and ground vegetation measured using cover estimates along transects in two Areas, 3 (8.7 ha) and 11 (11.2 ha). These Areas were chosen to represent trends on the western and south-eastern slopes respectively but it should noted that they were amongst the most vegetated parts of the island. In addition, the frequency and percentage cover of plant species in 15 permanent quadrats distributed across the island was recorded. Particular attention was also paid to tracking trends in the status of threatened species, including the palm *Hyophorbe lagenicaulis*, and non-native species.

Reptiles

For reptiles, direct standardised daytime counts of at least two Areas (Area Counts) in each sample year, plus timed searches at night (to give Encounter Rates) and transects (one each in Areas 3 and 12 - west and south-east slopes respectively - walked during the day and at night) were used. Visual counts provided indices of population density and were always underestimates. In 1996 mark-recapture sessions (10 in each of Areas 3 and 11) used to estimate population sizes indicated that in Area Counts between 5% and 55% of individuals were recorded, depending on the species. Reptile biomass was estimated from population estimates and mean weights of caught samples of all species.

Invertebrates

In 1989 and 1996, the relative abundance and taxonomic diversity of invertebrates was sampled in two gullies on the south-eastern slopes and two sites in Area 3 using pitfall traps (10 set for 24 h) and sweep nets (150 sweeps) during daylight. Catches were sorted to Order and the proportions of soft-bodied taxa preferred as prey by lizards (Arachnida, Dermaptera, Dictyoptera, Orthoptera, Lepidoptera, Diptera, Phasmida, Coleoptera) separated from non-preferred prey (such as Hymenoptera (mainly ants) and Isopoda).

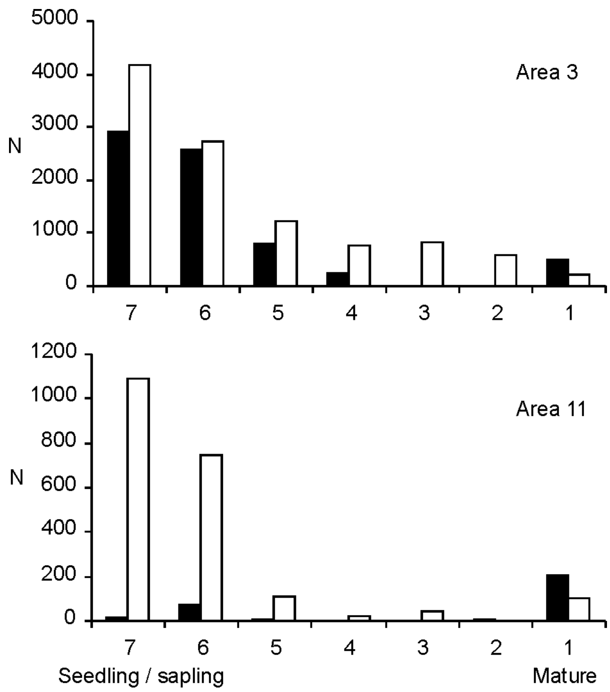

Fig. 1 Size structure of *Latania loddigesii* in Areas 3 and 11 of Round Island in 1975 (black bars) before browsing mammal eradication and 1996 (white bars) after browsing mammal eradication. Size class categories: 7, seedling; 6, >0.1 m - 0.6 m; 5, >0.6 m – 1.2 m; 4, >1.2 m – 1.8 m; 3, >1.8 m – 3.0 m; 2, >3.0 m, <2.0 m of trunk; 1, usually >3 m of trunk and 90 yrs. old. N = number of individuals.

RESULTS

Tree populations

Between 1975 and 1996 the number of Class 1 or equivalent individuals of the three main species declined. However, removal of the mammalian herbivores allowed large pulses of recruitment that are now beginning to replace losses of adult trees.

Latania loddigesii

Latania is by far the most abundant tree species on Round Island. After rabbit/goat eradication, the numbers of *Latania* in six smallest size classes dramatically increased. Eradication resulted in the first major recruitment phase to occur for at least 100 yrs. Between 1975 and 1996 the numbers of Class 1 *Latania* declined by 62% (Fig. 1). Assuming no change in the rate of decline, this cohort was estimated to disappear by 2009. If the mammalian herbivores had not been removed, by 2010 *Latania* would probably have disappeared, along with key habitats of several reptile species and a major component of the island's ecology.

In permanent quadrats, the density of *Latania* seedlings (Class 7) in the lower western, upper western and south-eastern slopes was similar within years but declined from a mean density of 8.2/m^2 in 1975 to 1.1/m^2 and 1.6/m^2 in 1989 and 1996 respectively. For quadrats on the lower western slopes, the decline between 1975 and 1989/1996 was significant ($F_{2,9}$ = 11.08, $P < 0.01$ on $\log_{10}(x+1)$ transformed data). As the quadrats were located in the areas of highest tree density, these results may therefore reflect localised self thinning.

Hyophorbe lagenicaulis.

In 1975 there were 15 mature palms on the island, only two of which were alive by 1996. However, recruitment increased markedly after 1986 with six newly-mature individuals in 1996, together with at least 42 young plants (over 1 m tall) and abundant seedlings in several locations. All these were the result of natural regeneration rather than planting as part of restoration programmes. Were mammalian herbivores still present, *Hyophorbe* would be close to extinction in the wild.

Pandanus vandermeerschii

On the south-eastern slopes, where *Pandanus* is most frequent, regeneration and recruitment increased after rabbit/goat eradication with, however, a decline in the numbers of trees in the oldest cohort (Fig. 2).

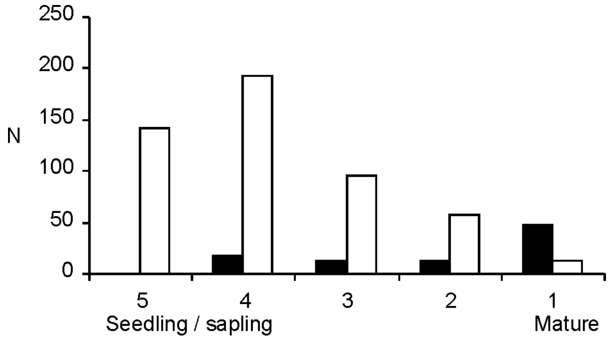

Fig. 2 Size structure of *Pandanus vandermeerschii* in Area 11 of Round Island in 1975 (black bars) before browsing mammal eradication and 1996 (white bars) after browsing mammal eradication. Size class categories: 5, >0.1 m – 0.6 m; 4, >0.6 m – 1.8 m; 3, >1.8 m – 3.0 m; 2, >3.0 m and vigorous; 1, >3.0 m showing signs of dieback. N = number of individuals.

Table 1 Changes in the ground vegetation cover in Areas 3 and 11 of Round Island between 1975 and 1996 based on point samples along transects. Goats were eradicated by 1978; rabbits were eradicated in 1986.

Vegetation/substrate type	% Cover Area 3				% Cover Area 11			
	1975	1982	1989	1996	1975	1982	1989	1996
Boulder, gravel, sand	21.1	5.9	3.0	4.5	3.7	2.8	5.5	7.1
Rock slab	62.8	73.3	63.3	56.9	41.8	37.3	37.2	33.8
Creeper	4.2	6.4	7.1	9.9	45.5	48.8	44.2	28.2
Exotic grasses	0.1	0.1	9.8	12.3	0.1	0	1.6	11.1
Native grasses	3.8	9.3	5.8	3.5	2.5	5.1	1.2	2.4
Tall herbs	0.1	0.1	1.8	2.8	6.6	5.1	8.5	9.8
Boerhavia spp.	0	0	0.9	1.0	0	0.1	1.4	2.8
Achyranthes	0	0	0	0.9	0	0	0	2.5
Latania	7.7	3.9	7.8	7.6	0.3	0.5	0.4	1.3
Pandanus	0.2	0.5	0.6	0.7	0.4	0.6	0.1	1.0
N point samples				3637				2942
Total vegetation cover	16.1	20.3	33.8	38.7	55.4	60.2	57.4	59.1

Table 2 The frequency and abundance of "important" plant species (species with >10% cover in any quadrat in any year) in 15 permanent quadrats on Round Island between 1975 and 1996. The number of quadrats in which the species was recorded (frequency) is followed, in parentheses, by the number of quadrats in which it was >10% cover (a measure of abundance). Species in bold are introduced (non-native). Responses in bold are major changes in frequency and/or abundance.

	1975	1982	1989	1996	Responses to eradication Goat	Rabbit
Passiflora	15 (8)	15 (3)	13 (7)	12 (6)	Unclear	Unclear
Tylophora	13 (5)	14 (11)	15 (11)	9 (5)	Increase	**Decrease**
Vetiveria	7 (5)	8 (5)	8 (4)	5 (0)	?Increase	**Decrease**
Portulaca	7 (3)	9 (0)	10 (0)	3 (0)	Unclear	Decrease
Ipomea	7 (0)	8 (2)	8 (5)	10 (7)	Increase	**Increase**
Ageratum	6 (2)	13 (0)	11 (1)	5 (1)	Unclear	Decrease
Nicotiana	6 (0)	7 (2)	0	0	?Increase	**Decrease**
Commelina	5 (1)	7 (1)	11 (3)	11 (1)	Unclear	Increase
Withania	2 (0)	0	0	1 (1)	Unclear	Unclear
Boerhavia	1 (0)	0	13 (0)	13 (2)	Unclear	Increase
Digitaria	1 (0)	0	6 (0)	7 (1)	Unclear	Increase
Chloris filiformis	1 (1)	1 (1)	1 (1)	1 (1)	Unclear	Unclear
Latania	0	1 (1)	6 (1)	7 (6)	?Increase	**Increase**
Abutilon	0	1 (0)	1 (0)	9 (7)	Unclear	**Increase**
Chloris barbata	0	0	10 (6)	13 (4)	None	**Increase**
Lycopersicon	0	0	1 (1)	0	None	Increase*
Amaranthus	0	0	2 (0)	5 (2)	None	Increase
Cenchrus	0	0	1 (0)	9 (4)	None	Increase
Achyranthes	0	0	0	13 (4)	None	**Increase**
Desmodium	0	0	0	2 (1)	None	Increase

***** Subsequently decreased

Ground vegetation

In Area 3, total vegetation cover along transects increased between 1975 and 1996, particularly between 1982 and 1989. In Area 11, where vegetation cover was much higher, there have been no substantial changes. In both Areas, a high percentage of the ground remains unvegetated a decade after rabbit eradication. In terms of changes in proportional abundance the most striking has been the increase in exotic grasses (*Chloris barbata, Cenchrus echinatus, Dactyloctenium ctenoides, Digitaria horizontalis)* and young *Latania.* Since the removal of rabbits/goats, vegetation dominated by *Boerhavia* spp., *Abutilon indicum* and *Achyranthes aspera* has increased, whilst native grasses (predominantly *Vetiveria arguta)* have declined. Creeper (*Tylophora coriacea, Ipomea pes-caprae, Passiflora suberosa*) cover increased in the more open Area 3 but declined in the more vegetated Area 11 (Table 1).

Non-native plant species

Of 72 vascular plant species recorded on Round Island since 1975, 37 (51.4%) are non-native (Appendix 1). The cumulative number of non-native species is increasing, but because of (re)discoveries of native species the native:non-native ratio has changed little. However, non-native species have had an increasing influence on the composition and structure of the ground vegetation. This effect is most marked for the present post-eradication period (Table 2).

The percentage of "important" species (see Table 2 for definition) which were non-native rose from 43% in 1975 to 75% in 1996. These non-native "important" species are those which are likely to be significant agents of change in the developing vegetation of Round Island.

The fate of individual non-native species on Round Island is hard to predict as shown in the breakdown of 37 species tracked to date:

Twelve species have become "important" components of the vegetation in one or more 15 permanent quadrats.

Four species have been "important" (*Lycopersicon esculentum* and *Nicotiana tabacum*) or common *(Tetragonia tetragonioides* and *Chenopodium murale)* but are now rare or have disappeared.

Six species are considered to be "potentially important" components. Three (*Cymbopogon excavatus, Solanum nigrum* and *Dactylotenium*) are common or frequent but have not yet reached >10% cover in any permanent quadrat. A further three (*Desmanthus virgatus, Heteropogon contortus* and *Chromolaena odorata*) have the potential to be invasive but to date have not

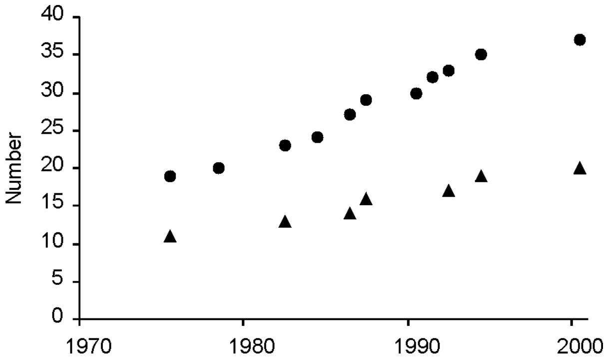

Fig. 3 The cumulative increase in non-native plant species recorded on Round Island between 1975 and 2000 based on Merton *et al.* 1989, North *et al.* 1994, Strahm 1994, pers. obs. The regression equations for all species (circles), and the subset of "potentially important" plus "important" species (triangles) are, respectively: y = 0.8165 – 1594.5, $F_{1,9}$ = 284.10, p <0.001; y = 0.3867 – 753.04, $F_{1,5}$ = 111.74, p <0.001. N = number of species.

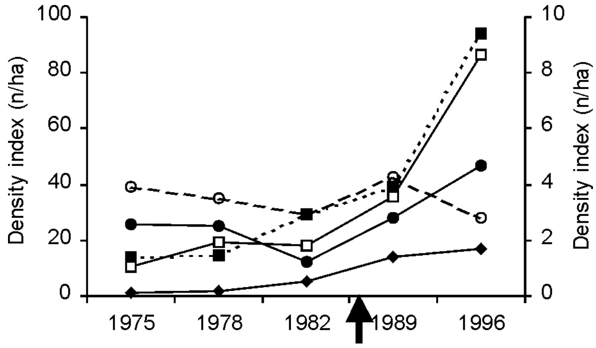

Fig. 4 Changes in the mean densities (n/ha) of the reptile species *Casarea dussumieri* (black diamonds, right axis), *Gongylomorphus bojerii* (white squares) and *Leiolopisma telfairii* (black squares), *Phelsuma guentheri* (black circles, right axis), *P. ornata* (white circles) recorded in daytime Area Counts on Round Island. Arrow denotes the year of rabbit eradication.

been recorded in permanent quadrats and are controlled by weeding or spraying.

Eight species have become (and remain) common but their influence on vegetation cover appears to be minimal and they are not believed to be significant agents of change.

Seven species which were never common have died out.

The number of non-native species recorded on Round Island increased linearly between 1975 and 2000. The rate of establishment increased after rabbit/goat eradication (Fig. 3) but not significantly so ($F_{1,7}$ = 0.83, p>0.05). For "important" species the rate did not change. Between 1975 and 2000 a new non-native species was recorded approximately every 1.4 years; a new "important" or "potentially important" species was recorded every 2.8 years.

Native plant species

Two native species, *Phymatodes scolopendria* and *Lobelia serpens,* have not been recorded since 1975. Changes in the populations of 15 further species which are largely or wholly confined to Round Island (Appendix 1) have been as follows: Five, *Latania, Hyophorbe, Pandanus, Gagnebina pterocarpa* and possibly *Lomatophyllum tormentorii*, increased in response to eradication of the mammalian herbivores. Four, *Dictyosperma, Fernelia buxifolia, Asparagus umbellulatus* and *Phyllanthus revaughanii*, have not responded and their populations remain critically low. Two, *Selaginella barklyi* and *Dichondra repens,* are low-growing and shade tolerant and remain widespread showing no obvious trend. Two, *Chloris filiformis* and *Aerva congesta*, remain confined to open and exposed habitats and their status has not changed. Two, *Vetiveria* and *Phyllanthus mauritianus,* appear to have declined since eradication.

Table 3. Comparisons of population indices of reptiles on Round Island. Comparison of Area Counts (day) and Encounter Rates (night) for before (1975, 1978, 1982) and after (1989, 1996) rabbit eradication. t tests used on $\log_{10}$ transformed counts. Transects, trends from counts in 1988 (from Merton *et al.* 1989), 1989 and 1996 are given for Area 3/Area 12 transects. '—' denotes species inactive.

	Area Count (day)	Encounter rate (night)	Transect Day	Transect Night
Phelsuma guentheri	No change	Increase P<0.01	No change/no change	No change/no change
Phelsuma ornata	No change	—	No change/decrease	—
Nactus serpensinsula	—	No change	—	Decrease/decrease
Leiolopisma telfairii	Increase P<0.01	Increase P<0.01	No change/increase	No change/increase
Gongylomorphus bojerii	No change	—	Decrease/no change	—
Casarea dussumieri	—	No change	—	Increase/decrease

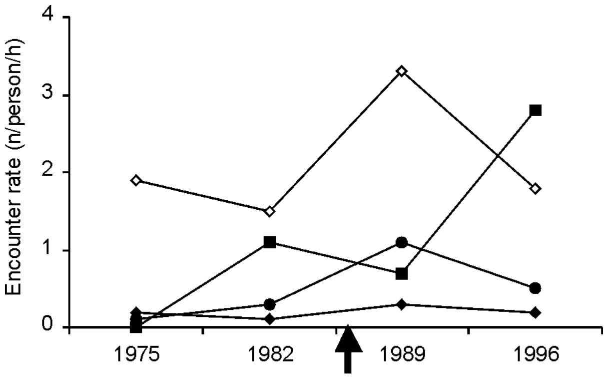

Fig. 5 Changes in the Encounter Rate (n/person/h) at night of reptiles on Round Island. Symbols as Fig. 4 but including *Nactus serpensinsula* (white diamonds). Arrow denotes the year of rabbit eradication.

Reptiles

Geckos

The large *Phelsuma guentheri* is now confined to Round Island where it has always been recorded as uncommon with a total population of <5000. Comparison of population indices from 1975 to 1996 indicates that overall this species has probably not increased in abundance, except at night when the Encounter Rate was significantly higher after rabbit eradication. Population indices of the nocturnal *Nactus serpensinsula* at night and the diurnal *Phelsuma ornata* during the day showed no evidence of sustained increases following eradication of rabbits (Table 3, Fig. 4 and 5).

Skinks

In contrast to the geckos, the large omnivorous *Leiolopisma telfairii* increased spectacularly after rabbits were eradicated. The small skink, *Gongylomorphus bojerii*, showed the same trend but the mean Area Counts from before and after rabbit eradication had high associated variances and the difference was not statistically significant (Table 3, Fig. 4 and 5). A third skink *Cryptoblepharus boutonii*, which has a wide tropical distribution, is largely confined to coastal rocks where its abundance does not appeared to have changed between 1975 and 1996.

Snakes

There is no evidence of a sustained increase in the abundance of *Casarea* as a result of rabbit eradication (Table 3, Fig. 4 and 5). However, the percentage of immatures found increased significantly from 22.0% in 1975 – 1982 to 50.5% in 1989-1996 (Chi square = 15.72, $p < 0.001$, df = 1) suggesting increased recruitment.

Relationships between vegetation and reptile abundance in 1996

The density of all *Latania* >1.8 m tall (Classes 5, 4, 3, 2 and 1) was a strong determinant of density of *P. ornata* (r^2 = 90.2%). Similar regressions for other reptile species showed much weaker relationships. Interestingly, the regression for trees and the arboreal *P. guentheri* explained less of the variation (r^2 = 41.4%) than did the one for the terrestrial *Leiolopisma* (r^2 = 51.4%). The latter is also strongly dependant on cover of litter, much of which was fallen *Latania* leaves (r^2 = 55.3%).

Changes in reptile biomass

Population estimates from between 1970 and 1975, based on tree:gecko ratios, mark- recapture and extrapolation from transects (Bullock and North 1976; Vinson 1975), indicate that reptile biomass on Round Island was relatively low at *c.* 4 kg/ha prior to goat/rabbit eradication. In 1996, using the same range of methods to estimate population sizes, biomass was approximately an order of magnitude higher at >40 kg/ha. Mean masses of reptile species had not changed (unpub. data), and the difference was principally due to an increase in *Leiolopisma*, together with increases in *P. guentherii* and *Nactus serpensinsula*. In 1996, *Leiolopisma* contributed 82.1% to the estimated reptile biomass.

Invertebrates

Comparison of catches in standardised pitfall and sweep net catches on Round Island in 1989 and 1996 showed no significant differences in either the diversity of invertebrate taxa sampled or number of soft-bodied "prey" species. Compared with other islands off Mauritius, such as Gunner's Quoin, samples from Round Island contained proportionately more soft-bodied prey. The unusually-high abundance of Dictyoptera on Round Island does not appear to have changed dramatically since 1975.

DISCUSSION

Responses of the tree populations

The imminent extinction of the oldest cohorts of *Latania* and *Hyophorbe* (and the decline in *Pandanus*) combined with negligible recruitment had mammalian herbivores still been present, indicates how close the palm-rich forest and its associated species had come to disappearing altogether. Increased regeneration and recruitment of woody species is a well-documented response to removal of grazing pressure. Despite the apparent adaptations of the three main tree species to deter herbivory, such as spiny seedlings and heterophylly, all were grazed by rabbits and/or goats. The recovery of the populations of *Latania*, *Hyophorbe* and *Pandanus* is the basis for restoring key processes such as accumulation of organic matter, soil retention and interspecific interactions. However, by 1996, recruitment of *Latania* seedlings in permanent quadrats had declined due to reduced seed bed availability, self thinning or interspecific competition. One or more of these factors is limiting future recruitment potential as predicted in North *et al.* (1994).

Responses of the ground layer vegetation

Three processes appear to be involved in changing the character of the ground vegetation post eradication, some of which were expected: 1. Observed declines in species of open, disturbed habitats that benefited from grazing pressure were expected. Such species are avoided by rabbits (e.g. *Nicotiana),* intolerant of competition (e.g. *Portulaca)* or tolerant of grazing (e.g. *Brachiaria* sp.). 2. The expected colonisation of bare areas has occurred but has been slow, because of the degree to which the island is subject to wind and the natural processes of sheet and gully erosion. 3. Where vegetation cover is high and the substrate stable, successional changes have occurred as expected. Annual species intolerant of shading (e.g. *Vetiveria, Portulaca, Nicotiana)* have declined whilst tall annuals/ biennials/perennials (e.g. *Abutilon, Solanum, Achyranthes, Boerhavia)* have increased. Non-native species have become an increasingly important influence on the structure and composition of the vegetation. This pattern has been described for other islands where introduced rabbits or goats have been eradicated (reviewed in Usher 1989; Zavaleta *et al.* 2001).

Observations suggest the following successional stages and their key constituent species (non- native species in bold):

a. Open vegetation, shade intolerant, grazing tolerant. Mainly annuals. *Vetiveria, Portulaca,* ***Nicotiana, Ageratum conyzoides, Passiflora,*** *Brachiaria, P. mauritianus.*
b. Closed vegetation, shade intolerant, grazing intolerant. Mainly annuals. ***Cenchrus, C. barbata, Digitaria.*** Potentially ***Dactyloctenium, Heteropogon.***
c. Closed vegetation, grazing intolerant. Low perennials including creepers. ***Boerhavia*** spp**.**, ***Commelina benghalensis, Desmodium incanum,*** *Tylophora.*
d. Closed vegetation of low scrub. Tall annuals, perennials, woody herbs/shrubs. ***Abutilon, Achyranthes, Solanum, Withania somnifera, Desmanthus,*** *Gagnebina.*
e. Palm/*Pandanus* thicket. Mature trees with thickets of younger trees. *Latania, Hyophorbe, Pandanus, Dictyosperma.*

Community type ‘a’ was extensive in 1975 but scarce by 1996 as ground cover and plant biomass increased. By 1996, ‘b’, ‘c’ and ‘d’ were widespread in vegetated areas but it is not yet clear how these changes will affect regeneration and recruitment of native tree species, especially *Latania,* which make up the characteristic landscape of Round Island and hold key habitats for the reptiles. The increasing influence of non-native woody herbaceous species has occurred on other Mauritian islands such as Ile aux Aigrettes (Dulloo *et al.* 1997) and was expected. However, only one woody shrub, *Desmanthus,* has become established on Round Island and that has been controlled by weeding. The effectiveness of control of *Desmanthus,* or other woody species, will be an important factor in the future development of the island’s vegetation.

Between 1975 and 2000 the cumulative increases in the numbers of new non-native species recorded on Round Island and the subset of “potentially important” plus ”important” species were linear (Fig. 3). This was despite wide fluctuations in mammalian herbivore density (in the case of rabbits from *c.* 2000 in 1985 to 0 in 1986), the number of human visitors (generally increasing over time) and the introduction of phytosanitary measures in 1986 (Merton *et al.* 1989).

Effects of eradication on populations of threatened native plant species

Five of the 15 native plant species which were threatened because of their induced restricted distribution and/or grazing, increased in response to the eradication of mammalian herbivores. These were all perennials and mostly trees. The remaining 10 showed no response. Either their populations were so low and they need a longer time to recover (and are perhaps genetically impoverished), or present conditions are unsuitable. Two species, *Vetiveria* and *P. mauritianus*, declined after rabbit eradication. Both are constituents of open vegetation community type ‘a’. In the absence of grazing, their persistence, along with that of two other species that thrive in open, salt-sprayed habitats, *C. filiformis* and *Aerva,* relies upon the intrinsic and natural instability of substrates, and exposure.

On Round Island there have been projected declines and losses of some key native plant species that can be directly attributed to the impact of mammalian herbivores. These, together with an apparent absence of impact on the rate of colonisation of non-native plant species indicates a differential “top down” control of vegetation. Goats, and especially rabbits, had much more overall influence on the presence or absence of native than non-native plant species.

Effects of eradication on populations of threatened reptile species

Except for *Cylindraspsis* sp. and *Bolyeria,* which were extinct or effectively extinct by 1975, no species appear to have declined. Of the threatened reptiles confined to Round Island, two types of response to the removal of rabbits were observed: Two species (*P. guentheri* and *Casarea)* do not appear to have increased. A third, *Leiolopisma,* has increased dramatically. Thus predictions of increases in arboreal reptiles (especially *Phelsuma* geckos) and possible declines in terrestrial reptiles have not been upheld. The estimated population sizes of *P. guentheri* and *Casarea* are in the low thousands and low hundreds, respectively, and these species remain vulnerable to extinction. Both are large, specialist reptiles with low reproductive rates; they are wholly or partly saurivorous, forage at night on the ground and also use above-ground vegetation. *Leiolopisma* is omnivorous and is seen with increasing frequency foraging at night. During the hottest part of the day it hides amongst litter. Its population appears to have benefited from the increased availability of litter produced as more *Latania* palms mature.

Reptile biomass is estimated to have increased ten-fold between 1975 and 1996 and is now approaching that of other seabird islands in the Indian Ocean such as Cousin Island, Seychelles. This change is mostly due to the increased population density of *Leiolopisma* between 1989 and 1996. In the absence of any known changes in the seabird populations the increase in *Leiolopisma* is attributed to increases in food or habitat availability after rabbit eradication. The abundance of invertebrates in 1989 and 1996 was similar, but neither pitfall traps nor sweep nets adequately sampled cockroaches (Dictyoptera), which are probably a major prey item for lizards that hunt at night, including *Leiolopisma*.

In the short term two skink species have responded positively to the ecological changes on Round Island induced by rabbit eradication. The high and apparently increasing population of *Leiolopisma* may serve to dampen increases in the other reptile species by way of predation and competition. This response of one native vertebrate species to the possible (short-term?) detriment of others has not been properly documented. It may represent an additional adverse trophic interaction to those already described for post eradication events (Zavaleta *et al*. 2001). In the longer term, density dependent factors may limit *Leiolopisma.* Increased availability of mature palms and *Pandanus* will provide more key habitat for *Phelsuma* geckos, and we expect the population of *P. guentheri* to respond positively to these changes.

Short-term and long-term consequences of eradication

Rabbits and goats were on Round Island for at least 150 years and their removal profoundly altered its ecology. The eradication of one of the mammalian herbivores (goat) led to a limited recovery (North and Bullock 1986). Whilst there are demonstrable benefits arising some 10 years after eradication of both goats and rabbits, we have also documented an increasing influence of non-native plant species. Of particular concern in the long term is the potential impact of non-native species on regeneration of tree species. In the context of Mauritius, Round Island is considered to have three key features: It is the best example of palm-rich forest (i.e. with a high proportion of native species); it has a large area of native vegetation; it is a refuge for species that now do not occur elsewhere. Notwithstanding eradication of goats and rabbits the increasing importance of non-native plants is reducing the value of the first two attributes. Furthermore, if tree regeneration is inhibited by non-native species then the value of the palm-rich forest as a refuge for threatened plants and reptiles may also decline.

All the data indicate that Round Island is experiencing a period of rapid changes. Some are successional, and the process is predictable; others, such as the rapid increase in the skink *Leiolopisma*, were not anticipated. Both types of response emphasise the value of continuing to measure the changes taking place on Round Island into the long term. Without further measurements of this "experiment" the ecological processes will not be understood in sufficient detail to inform management decisions on Round Island and similar islands elsewhere.

ACKNOWLEDGMENTS

The expeditions to Round Island would not have been possible without the support of the Government of Mauritius, (Ministry of Agriculture, Forestry and Natural Resources), The Mauritian Wildlife Foundation and The Jersey Wildlife Preservation Trust. D.J. Bullock and S.G. North thank The National Trust (UK) and Scottish Natural Heritage for their support. We also thank the hundreds of individuals and organisations that supported the expeditions in financial and other ways. They are individually acknowledged in expedition reports. J. Mauremootoo supplied new records of non-native plant species. We thank H.J. Harvey, A.S. Cheke, and J. Parkes (as referees) for critical comments on the manuscript.

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Appendix 1. Plant species list for Round Island. Observed abundance levels in 1975, 1982, 1989 and 1996. Common C, locally common L, frequent F, rare R, not recorded 0. *= Non-native species. # = Native species for which Round Island is of particular conservation importance (based on Merton *et al.* 1989; Strahm 1994; pers. obs.). Native/non-native status and nomenclature from Antoine *et al.* 1976

FAMILY	TAXON	ABUNDANCE	COMMENTS
Polypodaceae	*Phymatodes scolopendria* (Burm.f.) Ching	R O O O	Last seen 1975
Pteridaceae	*Adiantum rhizophorum* Sw.	F F F F	
	Pityrogramma calomelanos (L.) Link*	R R O O	Last seen 1986
	Pteris vitatta L.	O O R R	First found 1986
	Acrostichum aureum L.	R R R R	
Nephrolepidaceae	*Nephrolepis* sp. ? *	O O O O	Only seen in1986
Selaginellaceae	*Selaginella barklyi* Baker #	F F F F	
Thelypteridaceae	*Christella dentata* (Forsk.) Brownsey and Jermy	O O R R	First found 1986
	Thelypteris sp.?	O R R R	
Psilotaceae	*Psilotum nudum* (L.) Beav.	O O R R	First found 1987
Aizoceae	*Tetragonia tetragonioides* (Pallas) O.Kuntze*	C R O O	Last seen 1993
Amaranthaceae	*Aerva congesta* Balf.f. #	L L L L	
	Achyranthes aspera L. var. *aspera* L.*	O O O C	First found 1992
	Amaranthus viridis L.*	O O F L	First found 1986
Asclepediaceae	*Tylophora coriacea* Marais	C C C C	
Campanulaceae	*Lobelia serpens* Lam.	O O O O	Only seen in 1978
Caricaceae	*Carica papaya* L.*	R R R O	
Chenopodiaceae	*Chenopodium murale* L. *	C F F R	
Commelinaceae	*Commelina benghalensis* L.*	C C C C	
Compositae	*Ageratum conyzoides* L. *	C C C C	
	Bidens pilosa L. *	O O O O	First found 2000
	Chromolaena odorata (L.) King and Robinson*	O O O O	First found 2000
	Conyza canadensis (L.) Cronq. *	O O C C	First found 1986
	Crassocephalum rubens (Juss. ex Jacq.) S. Moore*	O O O O	Only seen 1990
	Eupatorium sp. *	O O O O	Only seen in1991
	Gamochaeta purpurea (L.) Cabrera * (formerly *Gnaphalium pensylvanicum* Willd.)	R R O O	Last seen 1982
	Sonchus asper (L.) Hill *	O O F F	First found 1984
	S. oleraceus L. *	F F R F	
	Tridax procumbens L. *	O O O F	First found 1991
Convolvulaceae	*Dichondra repens* J.R. and G.Forster #	F F F F	
	Ipomea pes-caprae (L.) R.Br.	C C C C	
Cyperaceae	*Cyperus rubicundus* Vahl.	L F F F	
	Fimbristylis cymosa R.Br.	L L L L	
Euphorbiaceae	*Euphorbia prostrata* Aiton *	? ? OO	Found 1978 and 1986
	E. thymifolia L.	F F F F	
	Phyllanthus amarus Schum. and Thonn *	O R F F	First found 1982
	P. mauritianus H.H. Johnston #	C C F R	
	P. revaughanii Coode #	O O R R	First found 1986
Gramineae	*Brachiaria* sp. (possibly *serpens* (Kunth) Hubbard)	C C R R	
	Cenchrus echinatus L. *	O O L C	First found 1987.
	Chloris barbata Swartz *	R F C C	
	C. filiformis (Vahl.) Poir.#	L L L L	
	Cymbopogon excavatus (Hochst.) Stapf *	R R R F	
	Cynodon sp (probably *dactylon* (L.) Pers.)*	R O O O	Last seen 1975
	Dactyloctenium ctenoides (Steud.) Lorch ex Bosser*	O O O F	First found in 1994/5
	Digitaria horizontalis Willd. var *porrantha* (Steud.) Henrard *	R R R F	
	Heteropogon contortus (L.) P.Beauv. ex Roem.& Schult. *	O O O R	First found 1994
	Lepturus repens (G. Forster) R. Br.	O O R R	First found 1989
	Sporobolus virginicus (L.) Kunth.	R R R R	
	Stenotaphrum micranthum (Desv.) C.E.Hubbard	R O R R	
	Vetiveria arguta (Steud.) Hubbard #	C C C C	

continued next page

Appendix 1. continued

Leguminosae (Mimosoideae)	*Desmanthus virgatus* (L.) Willd. *	O R L L	First found 1982
	Gagnebina pterocarpa (Lam.) Baillon #	O R R R	First found 1978
Leguminoseae (Papilionoideae)	*Desmodium incannum* DC. *	O O L F	First found 1987
Liliaceae	*Asparagus umbellulatus* Bresler #	O O R R	First found 1978
	Lomatophyllum tomentorii Marais #	L L L L	
Malvaceae	*Sida pusilla* Cav.	O O R R	First found 1986
	Abutilon indicum (L.) Sweet *	R F F C	
Nyctaginaceae	*Boerhavia coccinea* Mill.	F C C C	
	B. diffusa L. * (= *B. repens*)	R O F F	
Palmae	*Dictyosperma album* (Bory) H.Wendl. and Drude var *conjugatum* H.E.Moore and L.J.Guého#	R R R R	
	Hyophorbe lagenicaulis (Bailey) Moore #	R R R L	
	Latania loddigesii Mart. #	C C C C	
Pandanaceae	*Pandanus vandermeerschii* Balf.f. #	F F F F	
Passifloraceae	*Passiflora suberosa* L. *	C C C C	
Portulacaceae	*Portulaca oleracea* L.	C C F F	
Rubiaceae	*Fernelia buxifolia* Lam. #	O R R R	First found 1982
Solanaceae	*Nicotiana tabacum* L. *	C C C R	
	Physalis peruviana L. *	C F F F	
	Lycopersicon esculentum Mill. *	O L C R	First found 1982
	Solanum nigrum L. *	C F C C	
	Withania somnifera DC*	R F F F	

Number of species in each year of survey

	1975	1982	1989	1996	Total 1975-2000
Total	43	46	56	59	72
Native	24	26	33	33	35
Non-native *	19	20	23	26	37
Index of introduction (Usher 1988)	44.2	43.5	41.1	44.1	Overall = 51.4

Deliberately (re)introduced species (e.g. *Dracaena concinna, Argusia argentea*, *Scaevola taccada*, *Tarenna borbonica*) have not been included.

Introduced mammal eradications for nature conservation on Western Australian islands: a review

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Abstract There are about 3400 islands off the Western Australian coast, many of which have high nature conservation values. Eleven species of introduced mammals occur or occurred on 124 islands, including three domestic animals (horse, camel and sheep) that have not become feral. In addition, Aborigines introduced dingoes to at least four islands before European settlement. Six exotic mammals (red fox, feral cat, goat, rabbit, black rat and house mouse) have now been eradicated from more than 45 islands in a series of projects since the 1960s. Most effort has been directed at black rats with more than 31 islands now clear of this species. Pindone, vacuum-impregnated into oats, was used until the 1990s, when bran pellets with brodifacoum were used in the Montebello Islands. Rabbits have been eradicated using carrots soaked in sodium monofluoroacetate (1080), red foxes with dried meat baits impregnated with 1080 and cats with a combination of baiting and trapping. After a period of 20 years of ground shooting, goats were finally eradicated from Bernier Island using an experienced shooter operating from a helicopter. The house mouse has been eradicated from Barrow Island four times after introductions in food and equipment, and from Varanus and adjacent islands after introduction in food containers. Both islands are utilised by the petroleum industry. Difficulties and how they were overcome, and future eradication priorities, are discussed.

Keywords Exotic mammal eradications; rabbit; *Oryctolagus cuniculus;* goat; *Capra hircus;* house mouse; *Mus domesticus;* black rat; *Rattus rattus;* red fox; *Vulpes vulpes;* feral cat; *Felis catus;* sodium monofluoroacetate (1080); brodifacoum.

INTRODUCTION

Western Australia (WA) covers about one-third of Australia and has a correspondingly long coastline of about 12,500 km (Fig. 1). Most stretches of the coast are abundantly provided with offshore islands, islets and rocks, with only three long stretches of coast being island-free – the Great Australian Bight where the Nullarbor Plain meets the Southern Ocean, an area on the west coast adjacent to the Zuytdorp Cliffs between Kalbarri and Shark Bay and the Eighty Mile Beach between Cape Keraudren and Broome. If an island is defined as any feature above high water mark shown on a 1:100,000 map, there are 3424 islands in all (Department of Land Administration data, see Burbidge 1989). Most of these 'islands' are small with only 254 islands being larger than 100 ha; 90% of these are in tropical seas (Abbott 2000).

Most WA islands have very high nature conservation values (Burbidge 1989). These values include:

- the persistence of species of mammals now extinct or threatened on mainland Australia (Burbidge *et al.* 1997; Burbidge 1999);
- the presence of endemic taxa of mammals (Burbidge 1999), birds (Schodde and Mason 1999; Garnett and Crowley 2000) and reptiles (Cogger *et al.* 1993), many of which are listed as threatened, and of genetically unique populations of mainland species;
- the existence of examples of mainland ecosystems isolated by rising sea levels 14,000 to 6000 years ago that have evolved in isolation and that have not been affected by the exotics now widespread on mainland Australia;

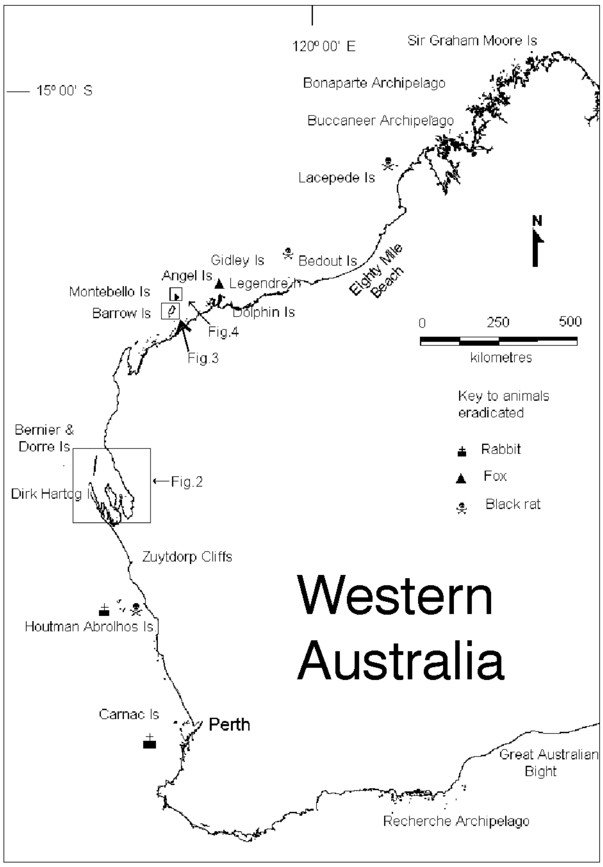

Fig. 1 Western Australian coastline showing islands where eradications have been undertaken. Detail of some islands provided in Figs. 2 to 4.

Pages 64-70 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Table 1 Introduced mammals and eradications on Western Australian islands

Feral mammal	Number of islands recorded	Number of islands eradicated	Number naturally extinct (or domestics removed)
Black rat*	40	31	-
Polynesian rat	2	-	-
House mouse	21	4	-
Dingo	4	-	-
Fox	8	4	-
Feral cat	17	2	5
Rabbit	14	6	2
Horse	2	-	2
Pig	1	-	-
Camel	1	-	1
Goat	6	1	1
Sheep	8	-	6
TOTAL	124	48	17

* Black rat also occurred on and was eradicated from 30-50 small islands (<15 ha) in the Montebello Islands.

- breeding sites for about 30 species of seabirds – in April 2001 the Department of Conservation and Land Management (CALM) Seabird Breeding Islands Database (Burbidge and Fuller 1996) contained 4821 breeding records of 42 species of 'seabirds' (as well as true seabirds the database includes other birds that breed on islands and depend on the ocean for their living) on 553 islands, and:
- nesting rookeries of four species of marine turtles (green turtle (*Chelonia mydas*), flatback turtle (*Natator depressus*), loggerhead turtle (*Caretta caretta*) and hawksbill turtle (*Eretmochelys imbricata*)) – most WA rookeries are on islands and many rookeries on the mainland are threatened by predation of eggs by the red fox (*Vulpes vulpes*) (Environment Australia 1998).

RESULTS

Islands with mammals

The Mammals on Australian Islands Database (Abbott and Burbidge 1995) demonstrates that at least 124 WA islands have or had introductions of 11 species of exotic mammals (Table 1). Most records are of black (ship) rats (*Rattus rattus*) (>40 islands), followed by the house mouse (*Mus domesticus*) (21), and feral cat (*Felis catus*) (17). Other recorded introductions are of Polynesian rat (*Rattus exulans*), red fox, European rabbit (*Oryctolagus cuniculus*), horse (*Equus caballus*), pig (*Sus scrofa*), one-humped camel (*Camelus dromedarius*), goat (*Capra hircus*), and sheep (*Ovis aries*).

In addition, dingoes (*Canis lupus dingo*) have been recorded on four islands (Augustus, Bigge, Middle Osborne and Wollaston) off the Kimberley coast. Dingoes were introduced to Australia about 3500 to 4000 years BP (Corbett 1995), well after island separation; therefore Aborigines, who in this part of Australia possessed limited seagoing capacity, presumably introduced dingoes to these islands.

Most islands off the WA coast south of the Kimberley have been reserved for nature conservation and are vested in the Conservation Commission of Western Australia and managed by the WA Department of Conservation and Land Management. The detrimental effects of exotic mammals on nature conservation values of islands are well documented (e.g., Atkinson 1985; Burbidge 1989, 1999, Burbidge *et al*. 1997) and the eradication of exotics from islands is an important Departmental role.

Case Studies

The eradication of exotic mammals for nature conservation purposes on WA islands commenced in the 1960s with a failed attempt to eradicate rabbits on Carnac Island near Perth using sodium monofluoroacetate ('1080') in oats. The first successful eradication was on the same island when, in May 1969, rabbits were eradicated using 1080 in fresh carrots (Morris 1989). Morris (1989) reported eradications from 1968 to 1985. All WA island eradications are summarised in Table 2 and eradication methods are summarised in Table 3.

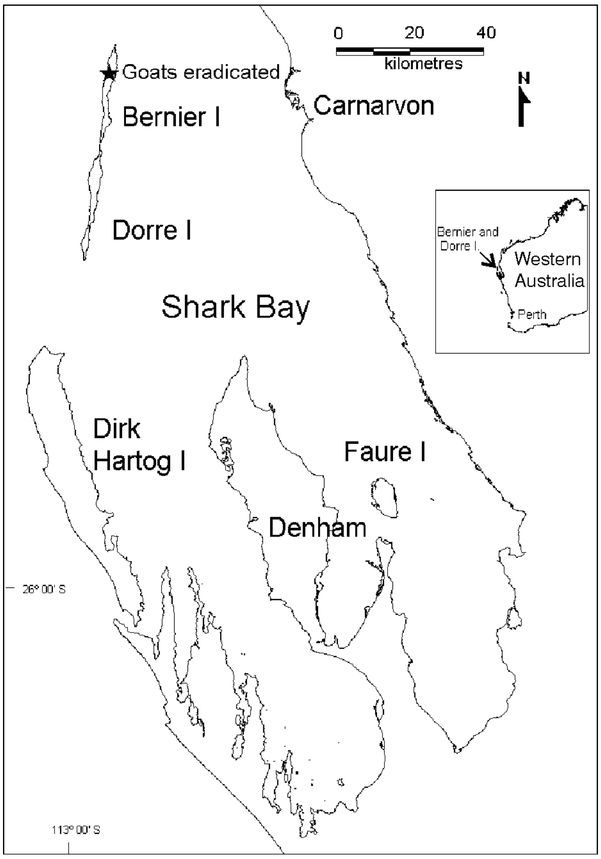

Fig. 2 Shark Bay area, showing Bernier, Dorre and Dirk Hartog Islands.

Table 2 Successful exotic mammal eradications on Western Australian islands 1969 to 2000

Island	Area (ha)	Year	Conservation values protected	Reference
Rabbit (*Oryctolagus cuniculus*)				
Carnac	19	1969	Breeding seabirds, vegetation	Morris 1989
Wooded	14		Breeding seabirds, vegetation	Morris 1989
Morley	7		Breeding seabirds, vegetation	Morris 1989
Leo	21		Breeding seabirds, vegetation	Morris 1989
Green Islets	6		Breeding seabirds, vegetation	Morris 1989
Goat (*Capra hircus*)				
Bernier	4267	1984	Threatened mammals, vegetation	Morris 1989
Black rat (*Rattus rattus*)				
Bedout	24	1981	Breeding seabirds	Morris 1989
Prince	4	1983	Adjacent to Barrow I	Morris 1989
Double	12 + 23	1983	Breeding seabirds, adjacent to Barrow I	Morris 1989
Boomerang	5	1985	Adjacent to Barrow I	Morris 1989
Pasco	2	1985	Adjacent to Barrow I	Morris 1989
Boodie	170	1985	Threatened mammal, adjacent to Barrow I	Morris 1989
West, Middle and Sandy, Lacepede Islands	82, 42, 6	1986	Breeding seabirds, turtle rookeries	R.I.T. Prince, pers. comm.
Barrow*	23 000*	1990/91	Threatened mammals	Morris (2002)
Middle (near Barrow)	350	1991	Threatened mammal, adjacent to Barrow I	Morris (2002)
Rat and adjacent islands (Houtman Abrolhos)	56	1993	None on island, possible invasion of nearby seabird breeding islands	Burbidge *et al.* unpublished
Montebello Islands (*c.* 180 islands, islets and rocks)†. Largest island (520 ha) to be used for mammal re-introduction/introduction	total >2000	1996, 1999	Breeding seabirds, turtle rookeries, islands	Burbidge 1997
House mouse (*Mus domesticus*)				
Barrow‡	23 000‡	1965, 1972, 1994, 1998	Threatened mammals, many other values	Butler 1970, 1985 CALM records
Varanus, Bridled, Beacon (Lowendal Islands)	80 + 22 + 1.2	1994-97	Breeding seabirds, vegetation	I. Stejskal, and J. Angus, pers. comm.
Red fox (*Vulpes vulpes*)				
Dolphin	3281	1980-1985	Native mammals	CALM records
Angel	927	1980	nature reserve	CALM records
Gidley	798	1980	nature reserve	CALM records
Legendre	1286	1980	adjacent to nature reserve	CALM records
Feral cat (*Felis catus*)				
Hermite (Montebello Islands)	1020	1999	Will allow reconstruction of vertebrate fauna	Algar *et al.* (2002)

* Eradication necessary over only 270 ha
† Eradication achieved on all but the largest island (Hermite 1020 ha)
‡ Introduced in food or equipment, eradicated before establishment

WA eradication operations present examples of the difficulties that arise and the ways that these are solved. Some examples are:

- Goats on Bernier Island. Goats were introduced to Bernier Island (4267 ha; Fig. 2) in 1899, when it was a pastoral lease. Initial attempts to eradicate goats during 1962-1972 involved shooters on the ground. Over 550 goats were removed during this period, but by the mid-1970s it became clear that the technique would never succeed as some goats could escape shooters by hiding in vegetation or in caves in cliffs. In 1984 funding became available to conduct helicopter shooting, utilising an experienced pilot-shooter team who had been involved in donkey control on the mainland. This proved a successful strategy (Morris 1989).

- Black rats on Barrow and Middle Islands (Fig. 3). Barrow Island (*c.* 23,000 ha) harbours many species of native mammals, including several that are listed as threatened, while the adjacent Middle Island (Fig. 3) has the threatened golden bandicoot (*Isoodon auratus*). Eradication required the development of a suitable bait station that excluded native species (Morris 2002).

Table 3 Eradication methods

Rabbit:	1080 in fresh carrots, three days of pre-baiting
Black rat:	Prior to 1996: Oats vacuum-impregnated with pindone on 25 m grid Post-1996: bran pellets with brodifacoum on 50 m grid, bran pellets with brodifacoum laid by helicopter
House mouse:	Varanus and adjacent islands: pindone-impregnated wheat and wax blocks with brodifacoum laid in bait stations on 20 m grid Barrow: bran pellets with brodifacoum
Red fox:	1080 in dried meat baits
Feral cat:	1080 in feral cat 'sausage' baits, followed by leg hold trapping
Goat:	Helicopter shooting

- Foxes on islands in the Dampier Archipelago. Red foxes self-introduced to Dolphin (3281 ha), Angel (927 ha), Gidley (798 ha) and Legendre (1286 ha; Fig. 1) sometime between 1930 and 1950 (Morris 1989). On Dolphin a population of Rothschild's rock-wallaby (*Petrogale rothschildi*) was near extinction by the 1970s, but remained abundant on nearby Enderby and Rosemary Islands, which did not have any foxes. Dolphin also has a population of the marsupial carnivore the northern quoll (*Dasyurus hallucatus*). In 1980, the use of dried meat baits with 1080 allowed the eradication of foxes without affecting the native carnivore, which has a significantly higher LD$_{50}$ (6.0-7.5 mg/kg cf. 0.13 mg/kg), even though the indigenous species has a lower body weight (fox *c.* 6 kg; quoll *c.* 250 g) (King *et al.* 1989). As well, dried meat baits are not attractive to the quoll. Re-invasion of foxes to Dolphin Island occurred in 1985 necessitating re-baiting. Since then Dolphin has been baited annually and the adjacent Burrup Peninsula has been baited four times per year. Monitoring of Dolphin has continued. The rock-wallaby population has recovered.

- House mouse on Varanus Island (Lowendal Islands; Fig. 3). In May 1993, the house mouse was introduced to Varanus Island (80 ha) in food containers supplied to an oil and gas base operated by Apache Energy. From there it spread naturally to nearby Bridled (22 ha) and Beacon (1.2 ha) Islands. Initial attempts by the company to eradicate near their facilities and then across all of Varanus Island using wheat with 1080 failed, probably due to insufficient bait being laid in bait stations that were too far apart and lack of follow up. After consultation with experts and better planning, eradication was achieved using wheat vacuum-impregnated with Pindone and wax blocks with brodifacoum laid in bait stations on a 20 m grid and maintained over a period of months. Eradication was achieved in 1997 (I. Stejskal, Apache Energy and John Angus, CALM, pers. comm.)

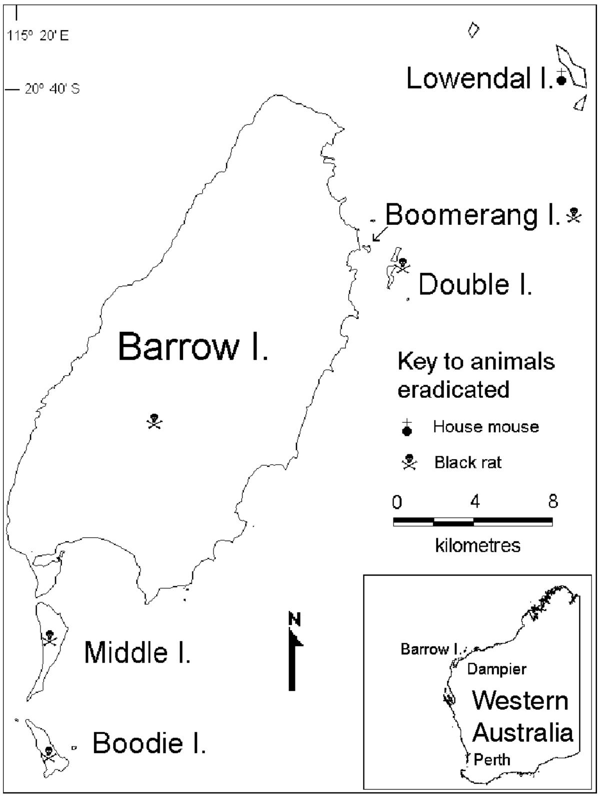

Fig. 3 Barrow, Middle, Boodie and the Lowendal Islands.

- Rats on the Montebello Islands. Black rats were introduced to the Montebellos (an archipelago of about 180 islands, islets and rocks totalling >2000 ha; Fig. 4) in the second half of the 19^{th} century (Burbidge *et al.* 2000). The presence of two granivorous birds (bar-shouldered dove *Geopelia humeralis* and brown quail *Coturnix ypsilophora*) required the development of a bait station that excluded these species and allowed access by rats. Experimentation on one island in 1995 showed that a bait station comprising a plastic bottle with two 43 mm holes cut in its sides provided a suitable method and in 1996 over 12,000 bait stations were installed and serviced on a 50 m grid on all islands. Eradication was achieved on all islands except the largest, Hermite at 1020 ha and two adjacent smaller islands, where rats were not detected until 1999 (Burbidge et al. 2000). No effects on the granivorous birds or on any other species, including raptors, were detected. A further eradication attempt, utilising bait laid from a helicopter spreader bucket, will take place.

DISCUSSION

The above operations provide a useful background to a discussion of eradication technology and issues.

Eradication, as opposed to control, is the desirable and possible outcome of operations against exotic mammals on islands. Parkes (1990, 1993), Bomford and O'Brien (1995) and Myers *et al.* (2000) have discussed eradica-

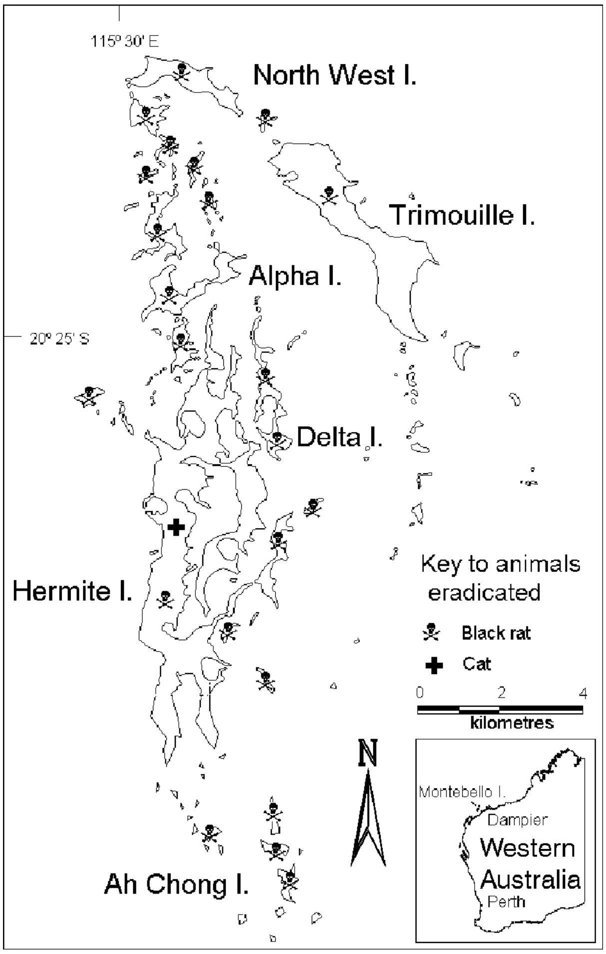

Fig. 4 Montebello Islands.

tion design and practice. Eradication on islands can be achieved if:

- the method used is capable of affecting all of the target animals over a short time,
- the method used kills or captures animals the first time they come into contact with the control method, so that bait or trap shyness does not occur,
- those carrying out the eradication have planned the operation to overcome all foreseeable obstacles,
- the method is applied until eradication is achieved, even if problems arise – financial planning needs to take account of possible initial failures and funding must be committed for the period needed,
- post-operational monitoring is carried out over an appropriate period of time, and
- programmes are in place to minimise the chance of re-invasion.

All the above are equally important. If a well-planned and implemented project fails a change in tactics may be needed – there is little point in repeating the same technique.

In addition, operations on islands with nature conservation values must take place with zero or acceptable impact on those values. Thus, the method used must be specific to the target species. Most operations on islands involve the use of poison in bait that is attractive to the target species. If there are no at-risk, non-target species present the technique can be used freely. If there are at-risk non-targets, the poison must either have no significant impact on them, or be presented in a way that makes it unavailable to them. In the latter case, experimentation may be required to develop bait stations that prevent access by non-targets. In some cases, where a very high conservation value species or community is being protected, some negative impacts may be acceptable if other alternatives are unacceptable. A further method, with limited application, is to remove the non-target animals, maintain them in captivity and return them after the bait is no longer effective. Where the possible effect of a technique is unknown, to where an established technique is proposed for use in a new environment, a pilot project/study may be required to ensure that non-targets are not significantly affected. That this can be the case is when brodifacoum was used in the Queen Charlotte Islands, Canada (Howald *et al.* 1999).

Various eradication operations in WA demonstrate the wide variety of problems presented and the ways that they can be overcome.

- The Bernier Island goat operation initially used a technique that was not capable of affecting all target animals in a short time – natural reproduction eventually equalled or became greater than the rate of population

Table 4 Future island eradication challenges for Western Australia

Red fox:	Depuch (1120 ha)
Feral cat:	Faure Island (5000 ha), Dirk Hartog Island (60,000 ha), Cocos-Keeling Islands (Home Island 100 ha, West Island 660 ha, South Island 390 ha)
Black rat:	Completion of Montebello Islands eradication, rats remain on Hermite 1020 ha, 140 km coastline)
Polynesian rat:	Eradication on two islands from which it is known, survey of other Kimberley islands
Pig:	Sir Graham Moore Island (2770 ha)

Table 5 Largest islands where eradication achieved for six feral mammals

Feral mammal	Largest Island.
Rabbit	Leo (Houtman Abrolhos) (21 ha)
Black rat	Trimouille (Montebellos) (520 ha) #
House mouse	Varanus (Lowendal Islands) (80 ha)
Fox	Dolphin (Dampier Arch.) (3280 ha)
Cat	Hermite (Montebellos) (1020 ha)
Goat	Bernier (4267 ha)

Eradication necessary over only 270 ha of 23,000 ha Barrow Island

reduction. Only when a better technique – helicopter shooting – became available, was eradication achieved.

- The Dampier Archipelago fox eradication shows that careful bait formulation can lead to eradication in the presence of a native mammal with a similar diet. It also shows that monitoring of re-invasion is needed, especially where the exotic remains nearby, and that a control or eradication operation may also be needed on nearby land.

- The Barrow Island/Middle Island rat eradication provides an example where the development of an appropriate bait station ensured that non-target mammals were not affected by eradication projects.

- The Varanus Island mouse operation provides an example where initial attempts to eradicate were poorly conceived, but where better design and a long-term commitment did result in eradication.

- The Montebello Islands rat operation is an example of the need to monitor post-baiting and to persist until eradication is achieved, despite a significant financial cost.

FUTURE NEEDS

Exotic mammals remain on several Western Australian offshore islands of nature conservation significance and further eradication operations will be needed. Table 4 shows some of the challenges that lie ahead. With the recent eradication of cats on Hermite Island (Algar *et al.* 2002), the technology to eradicate the exotic mammals concerned is available and tested in Western Australia for all exotic species except the pigs on Sir Graham Moore Island. However, some of the islands are significantly larger than the largest where eradication has so far been achieved (Table 5). In particular the eradication of goats and feral cats from the 60,000 ha Dirk Hartog Island will be a significant challenge.

WA islands are increasingly being used by the petroleum and aquaculture industries and for recreation. Quarantine procedures developed for Barrow Island by West Australian Petroleum Pty Ltd (now incorporated into Chevron Australia) have demonstrated that quarantine can be successful (Butler 1989). However, it has failed on at least four occasions when house mice have entered the island via food containers or in equipment. Even with the best will and efforts, quarantine can never be 100% successful and use of islands with high nature conservation values by industry should be minimised.

With boat ownership rapidly increasing, visits to islands for recreational purposes by Western Australians are also increasing. Monitoring of islands by conservation agency staff and others is needed to maximise the chance of detecting introductions as early as possible and response manuals are needed to allow staff to take appropriate and timely action.

CONCLUSIONS

Eradication of exotic mammals on WA islands of nature conservation significance has been achieved through a series of successful operations. Eradications are still required on several islands and steps are needed to prevent introductions continuing. There is an urgent need to develop monitoring protocols and a response manual for WA islands.

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Habitat refuges as alternatives to predator control for the conservation of endangered Mauritian birds

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Abstract Mammalian predators introduced to the island of Mauritius threaten the survival of several species of endemic birds. Long-term lethal predator control is achieving limited success against some predators but cannot be used against crab-eating macaques (*Macaca fascicularis*). Macaques are major nest predators on Mauritius, as confirmed by camera traps. Previous research suggested that plantations of non-invasive Japanese red cedar (*Cryptomeria japonica*) provide a refuge from nest predation. Using surrogate nests we show that nest predation by macaques is significantly lower in cedar than in native forest, including cedar plantations not currently occupied by rare native birds but which might be used for reintroductions. We present a simple habitat model showing how the careful planting and management of this non-invasive exotic, in conjunction with existing conservation efforts, could provide a sustainable solution to high predation rates by macaques.

Keywords nest predation; introduced predators; island ecosystems; predator control; refuges; invasive species; spatially explicit model.

INTRODUCTION

It is well known that animals that evolved in isolation on oceanic islands are hugely vulnerable to introduced, non-native, predators (King 1980; Atkinson 1985; Johnson and Stattersfield 1990; Moors *et al.* 1992). This is particularly evident on Mauritius, where predation by introduced mammals, in synergy with habitat destruction, has led to the extinction of at least nine endemic species of birds and reptiles (Cheke 1987). Predation of eggs, chicks and incubating adult birds (nest predation) is thought to be the greatest threat, posed chiefly by black rats (*Rattus rattus*) and crab-eating macaques (Safford and Jones 1998). In addition, feral cats (*Felis catus*) and lesser Indian mongooses (*Herpestes javanicus*) prey on adult birds (Roy 2001; C. G. Jones, pers. comm.). Consequently, black rats, feral cats and mongooses have for the last decade been controlled in areas where threatened endemic birds nest (Swinnerton *et al.* 1993). However, it is not clear whether control effort increases in direct proportion with the impact of a predatory species, and thus whether it is effective (cf. Côté and Sutherland 1997). Furthermore, eradication of most predators from Mauritius is not currently feasible due to its size, steep mountainous terrain and limited resources. Lethal control of macaques is anyway considered unacceptable, due to the socio-religious sensitivity of killing primates on Mauritius.

Predator control is now concentrated in conservation management areas (CMAs): fenced forest plots (1-23 ha; 2 m high galvanised steel fence) from which non-native rusa deer (*Cervus timorensis*) and wild boar (*Sus scrofa*) have been excluded and invasive non-native arboresent flora removed, permitting the regeneration of native forest trees (Strahm 1993). The latter are likely to provide more food for native birds (Safford and Jones 1998). On-going reintroductions of two endangered bird species, the pink pigeon (*Columba mayeri*) and echo parakeet (*Psittacula eques*) are centred around CMAs, where released birds are also provided with supplementary food. Despite all these measures nest predation continues to limit growth of native bird populations (C. G. Jones, pers. comm).

There is little quantified information on the impacts of different nest predators on Mauritius. Impacts of introduced rats have been well documented in temperate forests in New Zealand (Innes 1990; Moors *et al.* 1992), but little is known about them in insular tropical forests like those on Mauritius where rat density may be very high. Macaques are also numerous on Mauritius, with an estimated population of 40,000 (Bertram and Ginsberg 1994). They have long been suspected as important nest predators (Grant 1801), a suspicion reinforced by more recent authors (McKelvey 1976; Jones 1987; Safford 1991), although some have questioned their importance (Sussman and Tattersall 1986). Thus it is not clear which predators have most impact on native birds.

In a recent study of nesting success of the critically endangered Mauritius fody (*Foudia rubra*), black rats and macaques were suspected as the main predators from indirect evidence such as nest damage and eggshell fragments, although their relative impact remains unresolved (Safford 1997a). Importantly, Safford's study revealed that nesting success in introduced, but non-invasive, Japanese red cedar was significantly higher (46%) than in other trees (6%). Furthermore, the last remaining wild pink pigeons nested only in a cedar plantation ("Pigeon Wood"), one of the four mainland sites used for their reintroduction. This raises a paradoxical opportunity for conservation management: if introduced cedar provides a refuge, should it be more widely planted to reduce predation pressure even though it is non-native? Cedar plantations could provide a sustainable, strategic, alternative to lethal predator control and a means of reducing the impact of macaques, which cannot be controlled.

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In this paper we report on the value of automatic cameras and surrogate bird nests in identifying nest predators on Mauritius. We examine the *relative* impacts of different predators in potential refuge (cedar plantation) and other habitat types. We seek to extend Safford's (1997a) study by identifying the mechanism (e.g. relative predator density, habitat structure) that leads to some habitats experiencing lower rates of predation. We then develop a simple spatial model to suggest where new habitat refuges (cedar plantations) could be established to maximise benefits to endangered birds, without adversely affecting native vegetation. Our findings show that natural refuges from predation merit much more attention from conservationists attempting to combat the vast global impact of introduced predators.

METHODS

Study areas

We conducted fieldwork in the Black River Gorges National Park, Mauritius (Fig. 1). The National Park, established in 1995, covers 6574 ha and encompasses the largest tract of native forest on Mauritius (Page and D'Argent 1997). Work was concentrated in four areas: (i) Pigeon Wood and other nearby cedar plantations; (ii) Brise Fer; (iii) Bel Ombre; (iv) Combo (Fig. 1). The vegetation in the park ranges from heath and scrub to super humid upland cloud forest. Pigeon Wood (altitude 650-700 m) is in the latter zone and is the largest (6 ha) plantation of cedar that is surrounded by native forest. Brise Fer (300-650 m) is lower montane wet forest, dominated by *Diospyros* and contains a number of exceptionally rare endemic trees (Strahm 1993). Bel Ombre and Combo (200-400 m) are also lower montane wet forest. Bel Ombre is dominated by native trees such as *Labourdonnaisia glauca* (Page and D'Argent 1997), but has extensive plantations of exotic pine (*Pinus* spp.). Combo consists of lower-canopied native forest, heavily invaded by the exotic *Syzygium jambos*, and plantations of exotics including cedar. Brise Fer and Bel Ombre contain several CMAs.

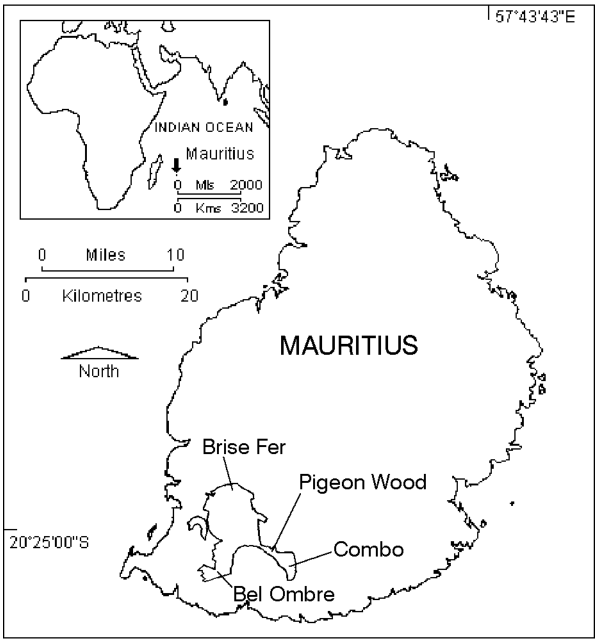

Fig. 1 Mauritius showing Black River Gorges National Park and the location of study areas.

Surrogate nests

We used disused nests of the introduced village weaver *Ploceus cucullatus*, secured with wire in trees with sufficient vegetation to conceal them. We fixed nests at 1.5-4.0m above ground (mean 2.23, SE 0.05), and baited each with a single domestic quail (*Coturnix coturnix*) egg and a similar sized model egg made of clay. Clay eggs registered the imprints of predators' teeth and so enabled the identification of predators that removed quail eggs without leaving other field signs. They also enable identification of smaller nest predators such as house mice (*Mus domesticus*) that are unable to open quail eggs (Roper 1992; Haskell 1995; Bayne *et al*. 1997) which average 30 x 25 mm (S. P. Carter, pers. obs.), but are probably capable of opening the slightly smaller eggs of some Mauritian passerines (e.g. Mauritius fody 18.4-19.9 × 13.0-14.6 mm) (Cheke and Jones 1987).

We minimised human scent left on nests and eggs by wearing rubber gloves during nest collection and preparation, and by rubbing our hands with mud and leaf litter during nest placement and checking (Reitsma *et al*. 1990). We prepared the clay eggs several weeks before use, thereby reducing any odour they might give off. Nests were classified as predated when one or both eggs were missing, broken, or tooth marked. We collected all nests at the end of the trial and opened remaining quail eggs to determine if they were still fresh.

Predator identification

We placed cameras fitted with a remote trigger mechanism modified from Major (1991) around 20 surrogate nests at the Fixon plot, a CMA at Bel Ombre. Photographic evidence thus obtained was used to confirm predator identity inferred from field signs (Fig. 2). Rats frequently consume eggs in or at nests, leaving characteristic 'boat shaped' eggshell halves with tooth marks around the shell edge (Fig. 3), and relatively large fragments embedded in the nest lining (Moors 1978; Safford 1994). They may also make small holes in the side of domed nests (Frith 1976). Macaques consume eggs whole, scattering a few small shell fragments in the vicinity of the nest (Safford 1994), and often tear domed fody nests apart (Jones 1987; Safford 1994). There are no published descriptions of mongoose nest predation, although they may occasionally climb trees and rob nests (S. Roy, pers. comm.). To take account of this we carried out feeding trials on captive mongooses.

Experimental design

A preliminary experiment conducted in 1997 outside the bird breeding season enabled us to assess how many nests would be needed and how long eggs should be left exposed in order to reliably measure predation rates. This experiment also provided a useful seasonal comparison of predation rates, although several of the CMAs were subject to varying intensities of rat control at this time. We placed up to 10 nests (mean 9.08, SE 0.53) at 25 m intervals, and at least 50 m from a vegetation type boundary, in three-seven replicate plots of three different vegetation types: (i) cedar; (ii) weeded native forest (CMAs); and (iii) unweeded native forest. Nests were left exposed for nine days and checked at intervals of three days.

We repeated the experiment during the 1998 breeding season (November-February) placing 10 nests in four replicate plots of the three vegetation types. During both experiments we ensured that no nest was visible to us from any of its neighbouring nests, and nest density (9/ha) was well below the recommended maximum for artificial nests (100/ha; Reitsma 1992).

Fig. 2 Photographs of nest predators taken with a nearby remote camera. Upper, crab-eating macaque; Lower, black rat. For trip cameras to function nests were placed in exposed positions and results from these nests were not included in the analysis.

Data analysis

Generalised linear models with binomial errors and logit link (GENSTAT®; Payne *et al*. 1997) were used to compare the proportion of nests predated in different vegetation types during the breeding season (S. P. Carter and P. W. Bright, pers. obs.). Statistical analysis was restricted to data collected during the breeding season at which time rat control was not being carried out in any of the native plots under study and only at low levels of intensity in one of the cedar groves. Separate models were derived for nests predated by rats and macaques. Minimum adequate models (Crawley 1993) were selected by first fitting all explanatory variables and first order interactions, and then testing the significance of each (using a χ^2-test statistic) by successive deletion.

Modelling the impact of habitat manipulation

A map of vegetation types in the National Park (Page and D'Argent 1997) was digitised into a geographical information system. We followed Page and D'Argent's vegetation classification, viz. grade 1: high quality forest, >70% indigenous vegetation; grade 2: moderate quality forest, 50-70% indigenous vegetation; grade 3: degraded forest, 20-50% indigenous vegetation; grade 4: highly degraded forest, <20% indigenous vegetation; non-native: monospecific or mixed species plantations of non-native trees. We then modelled the number of new fody territories that new cedar plantations could support, subject to the following constraints: (i) new cedar plantations could only replace non-native vegetation; (ii) new planting would only take place within the National Park and within the most recently documented range of the Mauritius fody (Safford 1998). We assumed that cedar plantations would support 100 pairs per km^2, as in and around Pigeon Wood, and that nesting success (i.e. the percentage of nests fledging at least one chick) was 46 % in cedar and 6% in all other vegetation grades (Safford 1997a, 1998).

Fig. 3 Nest contents from a rat-depredated nest. Eggshell fragment shown reveals that the quail egg was bitten along the longitudinal axis highly typical of rodent predation (Moors 1978). Rat incisor marks are clearly visible on the surface of the clay egg.

Mauritius fodies feed in native vegetation and nest in non-native trees only when there is native forest nearby (Safford 1998). Consequently cedar plantations remote from native forest might not be used due to the energetic costs of commuting flights between the two vegetation types, which we estimated as follows. We calculated total daily energy expenditure (DEE) for a passerine with the body weight of a fody (17.5 g; Cheke and Jones 1987), using the equation given by Walsberg (1983). Approximately half of DEE is required for general maintenance and at least 20% is directly expended in finding food (Walsberg 1983). We conservatively assumed that up to 25% of DEE was available to meet the costs of commuting flights during the nestling provisioning period. Flight costs per unit distance (kCal/km) were calculated using the equation for passerines derived by Kendeigh *et al.* (1977). We were thus able to determine the maximum total distance that could be flown by fodies expending 25% of DEE on commuting flights. Mauritius fodies have a relatively low provisioning rate of 2.8 feeds per nest per hour (Safford 1997b) as compared to other passerines of a similar size which make approximately eight feeds per hour (Williams 1987; Gill 1989). Allowing for twelve hours of daylight this amounts to 33.6 return nest visits per bird per day. Dividing total flight distance by the number of nest visits yielded the maximum distance between cedar plantations and native forest over which fodies could commute to forage.

Predation rates are likely to be higher where predators are more abundant, irrespective of potential habitat refuges. Macaque density is estimated to range from 0.33/ha in grade 1-2 forest to 1.3/ha in grade 4 forest and the mean troop home range size is 0.8 km^2 (Sussman and Tattersall 1986). We therefore incorporated a fourth constraint in our model: new cedar plantations must be >0.5 km from grade 4 forest, this distance being the radius of a macaque troops range in such forest. Black rat density appears to be high throughout the National Park, except where rats are controlled, and was thus not incorporated in the model.

RESULTS

Predator identification

The camera traps confirmed the validity of using previously-documented field signs to distinguish between different predators. In addition, rodent and macaque tooth marks in clay eggs enabled us to unambiguously identify the predatory species involved in 110 out of 122 cases of nest predation. Feeding trials confirmed that mongooses tended to carry eggs away from nests and broke eggshells into several large fragments and distinctive puncture marks from their canine teeth were often visible, however no nests were found to have been predated by mongooses in this study.

Black rats and crab-eating macaques were the only nest predators identified from photographs and field signs. At the end of the preliminary experiment 46 nests (34%) had been predated. Of these rats were responsible for 25 (54%), macaques for 17 (37%), and four (9%) could not be attributed to either predator. During the breeding season 76 nests (63%) were predated; rats were responsible for 37 (49%), macaques for 31 (41%) and eight (10%) could not be attributed to particular predators.

Predation rate and vegetation type

In a generalised linear model of the proportion of nests predated by rats during the breeding season, vegetation type was not included; predation rate did not differ between vegetation types (Fig. 4A). In a model of macaque predation, vegetation type explained 43% of the deviance; macaque predation was significantly higher in weeded (35%) and unweeded native forest (35%) than in cedar plantations (7.5%; GENSTAT, binomial errors P <0.001).

Outside the breeding season, vegetation type again affected predation by macaques. Predation by macaques was highest in unweeded native vegetation (37%), low in native weeded vegetation (9.2%), and absent from cedar plantations (Fig. 4B). Predation by rats was highest in cedar plantations (42%; similar to the rate during the breeding season) and low in both weeded and unweeded native vegetation (9.2% and 14.8% respectively). Figures 4A-B are not strictly comparable as several of the weeded plots and one

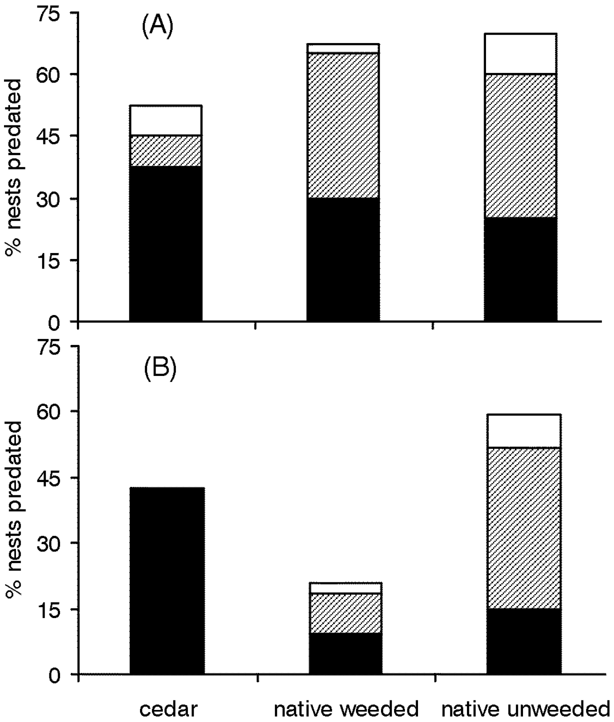

Fig. 4 Levels of predation in different vegetation types and during different seasons (A = breeding season; B = non-breeding season) on Mauritius from data collected using surrogate nests. Predators were black rats (solid bars) or macaques (striped bars) or unconfirmed (open bars). Sample sizes during the breeding season were 40 nests per vegetation type and outside the breeding season: 33 (cedar); 37 (native weeded); 27 (native unweeded).

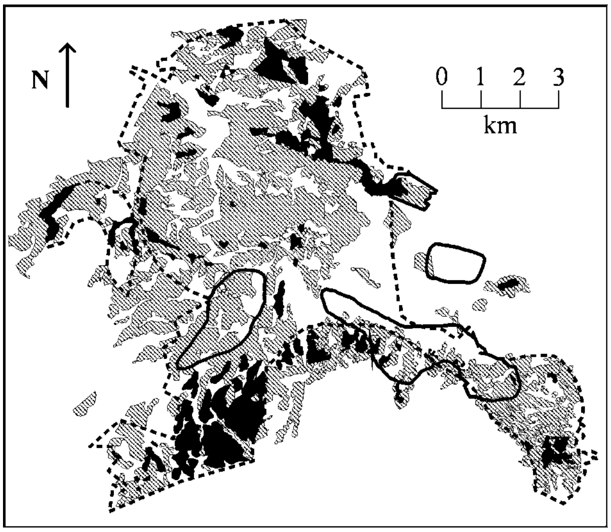

Fig. 5 Black River Gorges National Park (broken lines) showing extent of grade 1 and 2 forest (solid areas) and grade 3 forest (striped; see text for definitions), and last reported range of the Mauritius fody (solid lines; Safford 1998).

of the cedar groves were subject to varying intensities of rat control outside the breeding season; rat predation may have been higher in the absence of this control.

Siting of new cedar plantations

We estimated DEE of fodies to be 73.59 kJ/day and flight costs to be 0.57 kJ/km. Based on our model of 33.6 nestling provisioning flights per bird per day, this suggests a maximum distance of 1.02 km between new habitat refuges and native forest. However, fody territory size is seldom greater than 8 ha (Safford 1997b), therefore territory size is itself a constraint and, assuming a circular territory, new refuges would need to be within 320 m of native forest, this being the diameter of a circular territory 8 ha in area. There is approximately 5500 ha of native (grades 1-3) forest in and around the National Park (Fig. 5), and we calculated that there is 236 ha of exotic plantation (excluding cedar) within the National Park and present fody range. All of this is within 1 km of native forest, but if we assume that fodies cannot increase their territory size from 8 ha (see above) then the total amount of exotic plantation suitable for conversion is 192 ha. By imposing the additional constraint of increased macaque predation risk (i.e. new plantations should not be within 0.5 km of grade 4 forest), there are still 32 ha of exotic pine plantation that could be converted to cedar nest refuges (Fig. 6). The creation of each additional 5 ha of cedar could theoretically support five pairs of fodies, each of which may produce up to three broods per year (Safford 1997b). Based on Safford's (1997a) figures for fody breeding success, this could result in six new successful nests per year or one new successful fody nest for every hectare of cedar planted.

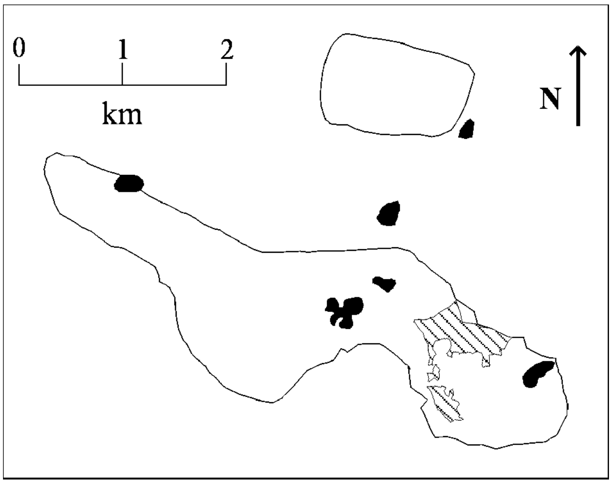

Fig. 6 Existing cedar groves (solid) within the vicinity of the Black River Gorges National Park and proposed areas for new cedar plantations (striped), in relation to part of the last reported range of the Mauritian fody (lines).

DISCUSSION

Surrogate nests and the measurement of nest predation

Deploying surrogate nests to estimate predation rates has three major advantages over observing predation rates on natural nests of rare wild birds: sample sizes can be much greater; nests can be positioned in habitats where relict populations may no longer nest, or perhaps have never nested; and no disturbance to nesting birds is caused. Surrogate nests provide a way to collect data of at least quasi-experimental rigour, which will often be of much more use in developing conservation management solutions than a few observations on individuals in a relict population. Such techniques have been criticised for lack of realism, but have usually involved the deployment of artificial nests which may not replicate real ones (e.g. Martin 1987; Picman 1988; Langen *et al.* 1991; Yahner 1991). In a recent review Major and Kendal (1996) showed that only 15 of 67 studies used nests constructed by the species under study or a similar species. In our study we used domed nests made by the village weaver, which closely mimicked those of the Mauritius fody (for nest descriptions see Crook 1963; Safford 1997c). We also very carefully concealed nests in live vegetation. There is thus every reason to expect that predators located and responded to our nests as they would to those of the Mauritius fody. The eggs we used were not intended to mimic those of the fody, since they would not have been visible to predators within the dark, domed weaver nests. With birds that construct cup nests, replicating the colour of eggs might obviously be important.

Despite the structural similarity between surrogate and fody nests, the former's paler colour may have made them more conspicuous to vision-oriented nest predators such as macaques. However, nest concealment was carefully quantified and analyses detected no association between macaque predation and nest visibility (S. P. Carter and P. W. Bright, pers. obs.). Furthermore, nest visibility did not differ between study plots or vegetation types. Olfactory cues might have rendered surrogate nests, constructed of unfamiliar grasses not necessarily found locally, containing quail and clay eggs and perhaps carrying human scent, more readily detectable by predators like rats with high olfactory sensitivity. Human visits to nests are known to result in heightened predation (Major 1990; Whelan *et al*. 1994). However, we visited nests infrequently and took considerable precautions to reduce human scent being left on nests or eggs. Furthermore, only two out of 134 non-predated quail eggs were putrid when cracked open at the end of experiments. Clay eggs have a distinct odour but Bayne and Hobson (1999) found that this neither attracted nor repelled mammal nest predators.

The lack of activity or odour from adult and fledgling birds might have made surrogate nests less conspicuous than real ones, balancing their possible heightened detectability from other cues. However whilst differences in detectability may exist between real and surrogate bird nests, there is no reason to expect the behaviour of predators foraging on surrogate nests to differ between different plots or vegetation types. Consequently, surrogate nests should provide a highly-reliable comparative measure of *relative* predation rates (cf. Martin 1987; Roper 1992; Yahner 1996; Penloup *et al*. 1997). They are also the only way to obtain estimates of predation in areas not currently used for nesting, but which might be important refuges or possible sites for future reintroductions.

Are cedar plantations a refuge from nest predators?

On average 7% of surrogate nests were predated per day during the bird breeding season. This is over 20 times higher than reported for a comparable study in Hawaii, where black rats were the only identified predators (0.33% per day; Amarasekare 1993). We found that both black rats and crab-eating macaques were important predators in Mauritius. These predators are abundant in the forests, which support virtually all remaining endemic bird populations (S. P. Carter and P. W. Bright, pers. obs.; D. Hall, pers. comm.). However, predation by macaques in cedar plantations was low, suggesting that cedar does provide a partial refuge from predation. This accords with Safford's (1997a) direct observations of lower (41%) predation of fody nests in cedar trees, compared to other tree species (82%). However it is important to note that we measured relative predation rates that cannot be compared directly to Safford's (1997a) measurements.

Safford and Jones (1998) suggested that predatory mammals may avoid cedar plantations due to lower food availability or because they are repelled by the resinous sap of cedar. The former seems the most likely of these explanations, since macaques were less abundant in cedar plantations (S. P. Carter and P. W. Bright, pers. obs.), yet climb individual cedar trees with apparent impunity (S. P. Carter, pers. obs.). We found no evidence that rats avoided cedar. Safford (1997a) also suggested that lower predation in cedar was the result of nests being more effectively concealed. However, concealment of surrogate nests did not influence predation by either rats or macaques. Lower predation by macaques is most simply explained by their lower abundance in cedar plantations.

Unsurprisingly, patterns of predation differed seasonally (cf. Safford 1997a). During the bird breeding season total predation was highest in unweeded and weeded native forest, and lowest in cedar plantations. Outside the breeding season, total predation was highest in unweeded forest and lowest in weeded forest plots (see below). Significantly, predation by macaques was lower in cedar than native forest both during and outside the breeding season. There is, however, a clear need for more information on the spatio-temporal dynamics of predator populations, which should greatly aid conservation management to reduce their impact on native ecosystems generally.

Predation in weeded native forest

Outside the bird breeding season total predation was much lower in weeded native forest. This might have been due to a behavioural 'fence effect' on the part of predators – the plots being surrounded by 2 m wire fencing – but neither rats nor macaques were physically excluded by the fence. Lower predation by rats at this time was almost certainly due in part to rat control being carried out at half of the weeded native plots. Perhaps as important was the lower cover of fruit-bearing shrub food sources, such as non-native guava (*Psidium* spp.), which had been removed from weeded plots and which fruits mainly outside the breeding season. In particular this may explain why predation by macaques was lower in the weeded plots at this time. As native canopy-forming trees mature in weeded forest plots, fruit-producing understorey shrubs may become less productive of fruits. Thus the restoration and maturation of native forest may at least partially help reduce predation – irrespective of seasonal effects – through reducing forest suitability for predators.

New cedar plantations as refuges from predation

Safford (1997a) showed that cedar plantations provide a refuge from predation for the Mauritius fody and anecdotal evidence suggests they are also a refuge for the pink pigeon. Using manipulative methods we have quantified the importance of different nest predators and have identified the probable mechanism leading to cedar being a refuge as lower macaque abundance in cedar plantations (S. P. Carter and P. W. Bright, pers. obs.). Cedar might thus be planted as a refuge for nesting birds from predation, and our model identified areas where this could be done without impinging on native forest which is under great threat (Lorence and Sussman 1986). The suitable areas are plantations of non-native pine and eucalyptus (*Eucalyptus robusta*) in and around the National Park.

A key requirement of new cedar plantations is that they are surrounded by or close to native forest where fodies and other native passerines feed. Our spatial model suggested that new cedar plantations would need to be very close (<320 m) to native forest. If plantations are to be away from highly-degraded forest where macaques are likely to be most abundant, there are 32 ha of exotic pine and eucalyptus plantations that could be converted to cedar. Relaxing this constraint, since we showed that cedar provides a refuge even when macaques are abundant, increases the amount of exotic forest available for conversion to 192 ha. Cedar nesting refuges could also be created outside of the present, highly-restricted, fody range, provided they were near to fragments of native forest. These could be used for reintroductions, following the approaches that have proven so successful with the pink pigeon and other endangered endemics (Jones and Swinnerton 1997).

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Control of invasive plants on the Poor Knights Islands, New Zealand

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Abstract In 1995 the New Zealand Department of Conservation initiated a weed control programme on the Poor Knights Islands, 16km offshore from Tutukaka, Northland, New Zealand. The intention is to eradicate all infestations of five environmentally invasive plant species (weeds) to the point where windborne re-invasion from seed sources on the mainland is the only threat. The invasive plants targeted, *Ageratina adenophora*, *A. riparia*, *Araujia sericifera*, *Cortaderia selloana* and *C. jubata*, are the only invasive plants present. All known weed sites are visited twice a year and all weeds found destroyed. Visits are timed to coincide with peak germination periods and pre-to-early flowering to prevent further seed set. Aerial surveys are completed during early flowering to locate any plants on cliff faces or in the canopy of trees. Areas of the island prone to reinvasion are thoroughly ground searched every year in spring, while the weed-free areas are searched every second year. *Ageratina adenophora* numbers have been reduced from several thousands to fewer than fifty. *Araujia sericifera* has continued to have high germination of seedlings, but is now in decline, probably because the seedbank is being depleted. *Araujia sericifera* has been the most difficult species to locate. A spreadsheet was developed that provides useful field data for control purposes and the raw statistical information for management and monitoring purposes. With refinement and manipulation this database could be beneficial for scientific research including species fecundity, seedling recruitment trends/time, seedbank viability under various geophysical site conditions, and rates of re-invasion from outside sources.

Keywords Mexican devil, *Ageratina adenophora*; mistflower *Ageratina riparia*; pampas grasses, *Cortaderia selloana, Cortaderia jubata;* mothplant, *Araujia sericifera.*

INTRODUCTION

Scope of this paper

This paper describes and discusses observations and actions at one location, with one group of target plants and the results of one management technique designed to fit the site conditions and plant behaviour in that site. It is a report on what was planned, how it was done and what has happened as a result. Analysis and comparison is left for others to consider.

Poor Knights Islands Management Area

Location and geography

The Poor Knights Islands (PKI) are situated 16km off the coast of Northland, New Zealand. The group comprises 272 hectares of land and consists of two main islands, Tawhiti Rahi and Aorangi, and seven smaller islets. The islands themselves are a Nature Reserve and are administered by the New Zealand Department of Conservation (DOC). They are surrounded by an 800m-wide Marine Reserve which is internationally recognised for recreational diving. The general public are not allowed access to the islands and all landings are by permit only.

The islands were created by ancient volcanic activity and have been geographically separate from the mainland for longer than any other islands around New Zealand's immediate coast. Thus the Poor Knights biota has one of the highest rates of local endemism in New Zealand (Nieuwland 1999).

History of human contact

Maori, the Polynesian settlers of New Zealand, had significant settlements on both the main islands until a massacre occurred around 1820. The islands were then declared sacred and settlement ceased. Prior to the massacre, Captain Cook, an 18th century English explorer, gifted pigs *(Sus scrofa)* to Maori on Aorangi island, and when the Maori left these animals reached high numbers (Fraser 1925). A successful feral pig eradication project was completed in the 1930s (Challies 1976). No other mammals have been recorded.

Invasive naturalised plants targeted

There are numerous alien plant species on the islands but only five have been identified as likely to have adverse impacts on the islands if left uncontrolled. These are Mexican devil (*Ageratina adenophora*), mistflower (*A. riparia*), two pampas grasses (*Cortaderia selloana* and *C. jubata*) and mothplant (*Araujia sericifera*). These weeds invade open disturbed sites, forming dense swards that outcompete the native regeneration. They gradually expand their range from the margins of their infestations by encroachment and displacement as the surrounding native species die out. Mistflower is shade-tolerant, so it can penetrate the forest interior and smother native seedlings. Mothplant seedlings are also shade-tolerant and remain in a phase of low foliar growth, until conditions such as increases in light and moisture levels enable a burst of growth up into the canopy.

All five species were introduced to New Zealand as garden ornamentals around 1900. By the 1930s they had natu-

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ralised and started spreading. A trypetid gall fly (*Procecidochares utilis*) was released in 1958 as a biological control agent for Mexican devil (Hoy 1960) but it does not provide successful control. To control mistflower the white smut fungus *Entoloma ageratinae* was released in 1998 (Frollick 1999) and it appears to be very successful. These control agents have not been observed on the Poor Knights although they are present on the Hen and Chickens Islands which are a similar distance from the mainland. The control agents are certainly capable of reaching the islands so their absence is probably due to the Poor Knights now having a lower weed density and therefore less chance for the control agents to establish.

Determination of when the invasive weeds arrived on the islands is difficult. Pampas grass, Mexican devil and mistflower have been widespread in Northland (Fig. 1) since the 1950s. Mothplant appeared in Northland around the 1980s. Pampas grass was first recorded on the Poor Knights Islands in 1974 (Veitch 1974) and Mexican devil in 1986 (Daugherty and Powlesland 1986). Mistflower was first recorded in 1991 (Wright 1991) and mothplant in 1993 (Parrish 1993). Density or distribution was not recorded at these early stages. Seedbank longevity of these species under New Zealand conditions is unknown.

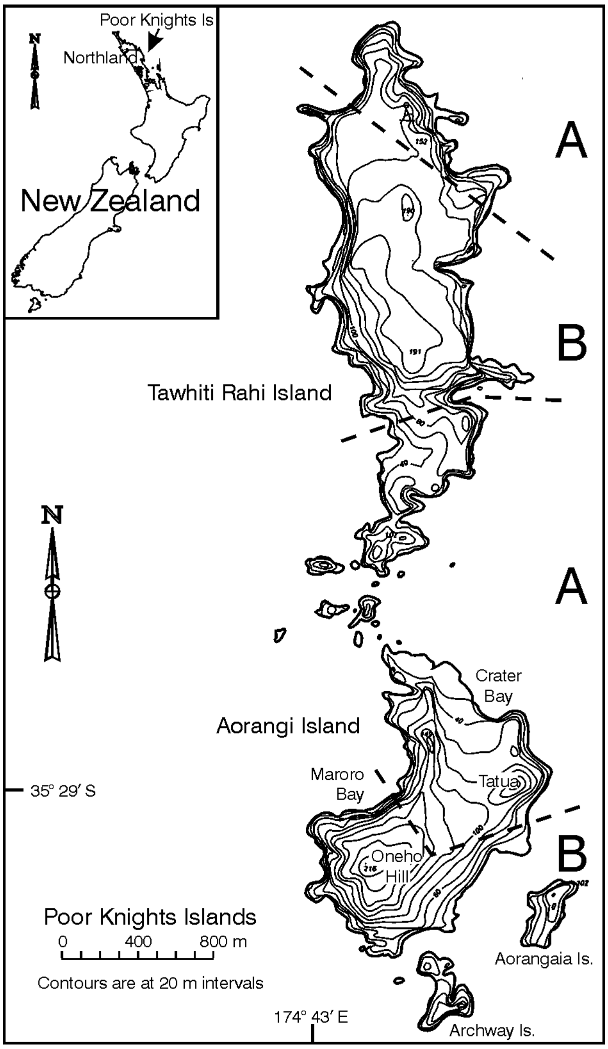

Fig. 1 The Poor Knights Islands showing their relationship to the coast of Northland and the places named in the text. The weed areas are marked as: "A" weed-free prone to invasion – contains known sites; "B" weed-free not prone to invasion – no known sites.

METHODS

Early control efforts

Control of pampas (not identified to species level at this time) commenced in 1991 and focused on three obvious infestations on the coastline. Control of the other three invasive species commenced in 1994. This work was done using volunteers and involved one trip a year to each island. No formal search techniques or data recording protocols were in place and the information gathered from sites was lacking detail or extremely variable. Sites were marked with various techniques. Random searching patterns at this point also resulted in many sites not being found. By 1995, 36 sites had been located.

Weed Eradication Strategy Poor Knights Islands (WESPKI)

A formal Weed Eradication Strategy for the Poor Knights Islands (referred to as WESPKI) was developed in 1996 (Bowden and Bowden 1996). The purpose of the strategy was to give control direction for the following five years and to standardise procedures for all weeding teams and data collection. The sensitive nature of the cultural and ecological values of the islands were also recognised. There was little information available at this time regarding the individual weeds and their attributes in this type of environment. Suitable techniques for intensive survey, relocation of sites and eradication were not available. The strategy was developed around errors uncovered during early control efforts, relating to seasonal timing of visitation and an appropriate site management regime.

This strategy has been altered each year since 1996 to recognise newly-developed best practices. Further reviews will continue to redirect it for the next five years. When it was developed, WESPKI was referred to as an "almanac for island visitation." Factors such as site hygiene and minimising impacts of visitation were incorporated because they impact directly on the efficacy of the weed work and success of the programme. Key aspects of WESPKI are listed below:

Management regimes and island visitation

The islands have been divided into three management zones: actual weed sites; weed free zones prone to invasion; weed free zones not prone to invasion. These were determined with consideration to the proximity to existing weed sites and the type of vegetation cover present. Where unmodified pohutukawa *(Metrosideros excelsa)* forest is found there is a striking absence of invasives. All existing weed sites occur in areas of disturbance associated with exposed coastal faces, shrubland and broadleaf forest. The weed free areas not prone to re-invasion coincide with the major seabird breeding areas.

A weed site is defined as any continuous infestation of weeds usually defined by proximity to a mature plant for mothplant or by the size of the light patch in which the weeds are present. Weed sites range from 1m x 1m plots with an individual plant to areas 20m x 30m. These are searched twice a year in early spring and early summer to coincide with the earliest flowering to occur.

Weed-free zones prone to invasion consist of the northern and southern tips of Tawhiti Rahi and Crater Bay, Tatua and Maroro Bay on Aorangi, and incorporate all those areas surrounding existing sites. These are intensively searched annually in early Spring.

Weed-free zones not prone to invasion include the seven islets, the tableland on Tawhiti Rahi and Oneho Hill on Aorangi. These are searched every second year in spring.

Every spring four people spend two days transit/setting/breaking camps, four days searching and visiting sites and weed-prone areas on Aorangi and three days searching and visiting sites and weed-prone areas on Tawhiti Rahi. On each alternate year half of the weed-free areas not prone to invasion are searched. In summer sites are re-checked by four people spending one day transit/setting/breaking camp, three days on Aorangi and one day on Tawhiti Rahi.

Aerial surveys

An aerial search is conducted annually in early summer for flowering mothplant and pampas grass and, every second spring for flowering mistflower and Mexican devil. In the case of mothplant it is virtually impossible to locate mature vines from the ground in dense vegetation once the plant has reached the canopy. Some coastal faces are not accessible by foot and the only way to search them is by aerial observation.

Database and information recording

The information recorded provides detail for re-location of weed sites and for analysis of the success of the weed programme.

For new sites details are taken describing the location and size of the site and satellite infestations. Details for relocation from other sites or from the track system are recorded. For both new and re-visited sites details of weeds removed include: date; species; numbers of adult/immature; control actions taken; and team leader. Adult plants are those which have completed a cycle from germinating to setting and dropping seed.

Search techniques

Sweep searching is conducted during spring visits to locate new infestations. Except for mothplant the species are all just commencing flowering at this stage. The sweeping technique involves all team members. They space themselves 10-20m apart, dependent on terrain and visibility, and move in line abreast between reference points. When weeds sites are encountered all weeders come together to record and intensively ground search the site. They then spread back out and continue sweeping. Intensive ground searching of sites by people on their hands and knees will pick up the majority of seedlings which could set seed by the next visitation. During the summer visits only known sites are visited. This avoids accidental movement of seed and disturbance of breeding seabirds.

Search timing

In northern New Zealand Mexican devil and mistflower can set seed from mid-spring to late summer although peak seed set is around late spring. The first treatment is therefore timed for early spring prior to seed setting in late September/October. During spring/early summer the time taken from germination to maturity is much faster than over autumn/winter and January has proven to be the best time to revisit sites to catch plants that have germinated since the spring trip, prior to their setting seed in late summer (February/March).

Site marking protocols

All weed sites are marked with a purple plastic triangle with the site number written on it. This is placed in the centre of the site with a piece of pink flagging tape to identify its location. Around the boundary of the site more pink flagging tape is installed also with the site number and reference to its position on the site (e.g. northern limit of site 124). The labels are replaced regularly to avoid perishing completely. The site location is recorded on GPS (Global Positioning Systems).

Weed removal

All flowers and seedheads are removed from the plants and placed in secure bags for removal from the island.

All plants are hand pulled, soil is shaken from the roots and the plant is placed so that the roots are clear of the ground. The roots of larger mothplants are grubbed out to ensure that they do not re-grow.

In the first two years of this operation adult mothplant stems were cut and painted with a herbicide mix of metsulfuron methylester (600g/kg) at 2g/2*l* of water.

RESULTS

Field trips in 1996 required 96 person/days per year: 40 person/days searching and 56 person/days weeding sites. It now takes 56 person/days per year: 40 person/days completing surveillance for new infestations and 16 person/days searching and controlling the existing sites.

On the Poor Knights Islands 142 weed sites have been recorded since 1995. During the visit of 9 February 2001,

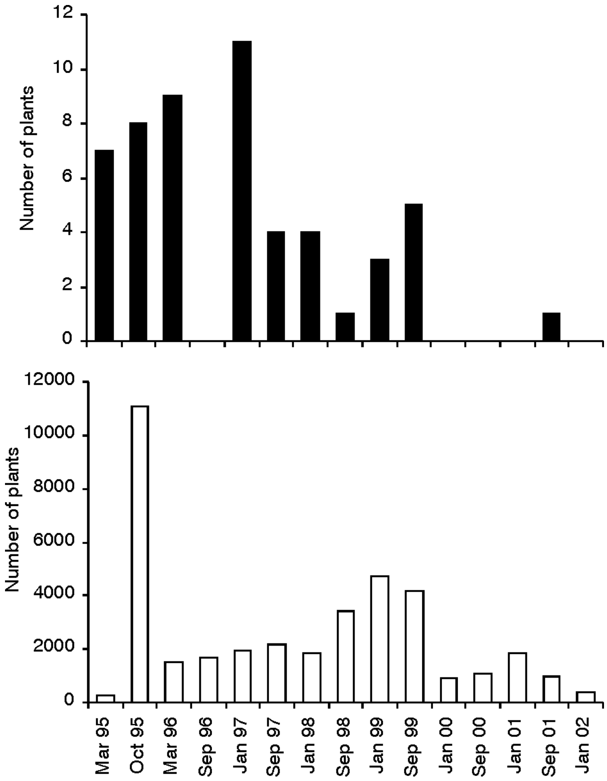

Fig. 2 The numbers of mothplants (*Araujia sericifera*) destroyed on Aorangi Island. The upper graph is adult plants and the lower graph is immature plants.

112 of these were weed free. A comparison of the detail recorded in our field trip reports over successive years, shows that the number of clean sites continues to increase and the number of new sites has rapidly decreased.

The numbers of weeds controlled on successive visits are presented in Fig. 2-5. There are seasonal fluctuations between spring/summer visits but there is a clear declining trend. The actual 'effort' in terms of applying attention to thoroughness has remained constant during searching within sites for weeds and during surveillance for new infestations. As there are now fewer weeds to remove the hours required to complete trip visits have reduced.

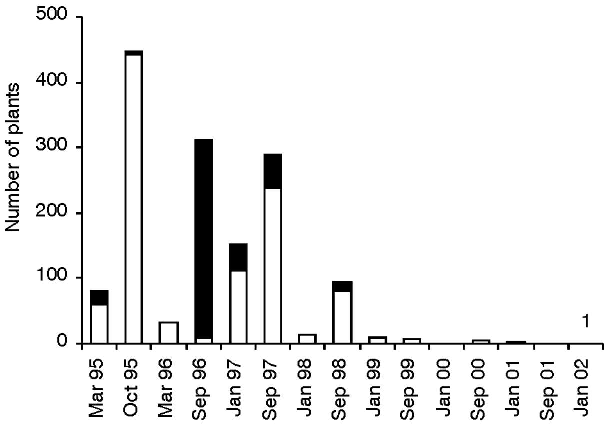

Fig. 3 The numbers of mistflower (*Ageratina riparia*) plants destroyed on Aorangi Island. The black bars are adult plants and white bars immature plants. Note that one immature plant was destroyed in Jan. 2002.

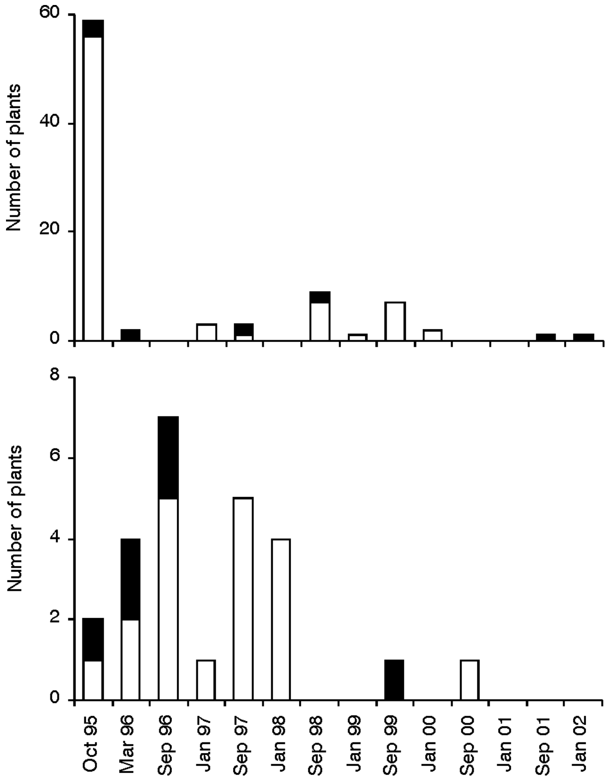

Fig. 4 The numbers of pampas plants (*Cortaderia selloana* and *C. jubata* combined) destroyed on Aorangi Island (upper graph) and Tawhiti Rahi Island (lower graph). The black bars are adult plants and white bars immature plants.

The total number of mothplants (Fig. 2) has fluctuated as new sites were found after aerial survey commenced in January 1997. However, since this time the trend line has started a steady decline, although this appears to be slower than other species and could be a reflection of greater seed longevity in the soil.

Mistflower (Fig. 3) showed an interesting reversal in adult to juvenile plants numbers around 1996/1997. This came about because two mistflower sites were missed and were full of adults the following year. This stresses the importance of visiting sites at least twice a year to beat the setting of seed. After this, juvenile numbers increased and then a steady decline occurred as the seed bank started depleting.

Pampas grass control commenced on Aorangi earlier than on Tawhiti Rahi (Fig. 4). There has been a dramatic decline in numbers of pampas after the large infestations were dealt with. Pampas probably has a shorter seed viability than the other weeds.

The reduction of mature Mexican devil has been similar on both islands (Fig. 5) but immature plants continue to occur on Aorangi. There have been significant benefits

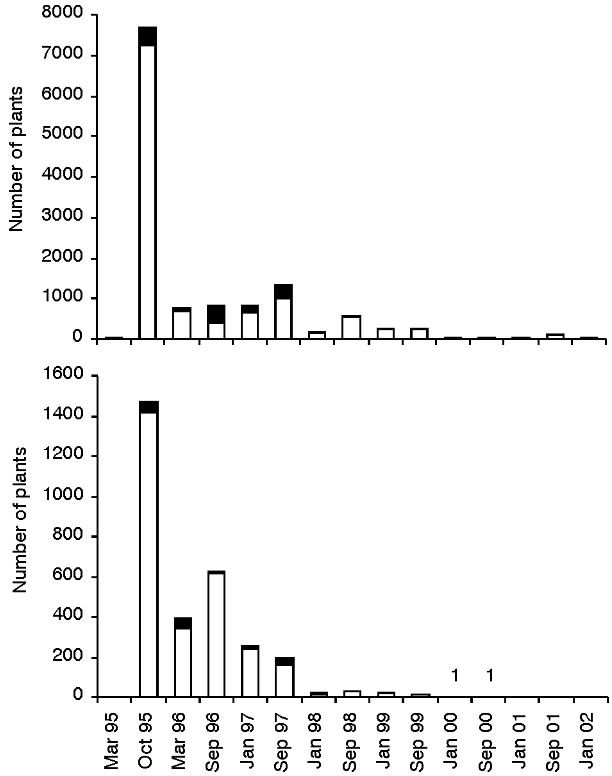

Fig. 5 The numbers of Mexican devil (*Ageratina adenophora*) plants destroyed on Aorangi Island (upper graph) and Tawhiti Rahi Island (lower graph). The black bars are adult plants and white bars immature plants. Note that one immature plant was destroyed on Tawhiti Rahi in Jan. 2000 and one in Sep. 2000.

from commencing control on Tawhiti Rahi before the weed was well established.

Tawhiti Rahi had no weeds during the visit of 9 January 2001 and consistently had only one or two plants found during each of the last four visits. There have been no new sites found for more than two years. The fortunate factor that has resulted in success is the simple fact that weeds had only just started to establish and never reached significant populations. It is my assumption that these sites may have established later from seed dispersing from Aorangi.

Mist flower on Aorangi had declined to zero plants but then two new sites containing one plant each in the first year of flowering (one of which had set seed) were located. Mexican devil, which was the most widely dispersed and prevalent weed, is now down to around 20 plants, all juveniles, found during each search.

DISCUSSION

Although control was not commenced until well after the invasive species had arrived on the islands, we started control before massive encroachment had occurred. In any eradication programme involving these five species this should be a fundamental criterion in evaluating whether or not to proceed with eradication or focus on sustained control. Once populations have become well-established, the ability to achieve coverage over all sites at the critical management times makes it difficult to beat the rapid cycle from germination to seeding.

A team of four people was found to be a good number to manage. More people become difficult to keep in formation and resulted in delays waiting for others to catch up. Fewer people meant sweep searches were narrower and took longer to complete. For greater efficiency the team can split in two to deal with very small sites less than nine metres square.

Existing literature on the topic of weed control consists either of site-specific data, general autoecological/biological information about species, or models on plant behaviour such as dispersal. The behavioural traits of species vary dramatically in different geophysical and climatic contexts. For example, *Ageratina adenophora* is also invasive in Australia and Hawaii, yet in all three locations it occupies very different habitats and has different behavioural characteristics, such as seed density and infestation/proliferation density, compared with its growth in New Zealand. The techniques described here would probably be excessive in Hawaii and inadequate in Australia.

Seed viability

We do not know the maximum time over which seeds can remain viable for any of the species being controlled in this environment. Individual site trends on the Poor Knights suggest that significant seed bank depletion for pampas grass, mistflower and Mexican devil occurs between two and four years. Mothplant seeds seem to persist for longer as seedling numbers only started to decline after five years. Mothplant does not show a decline as clearly because a number of adults have been located in the last two years. It is expected that they may persist for ten years. We initially planned to declare individual sites weed-free and to "archive" the site (i.e. no longer specifically search the site twice a year) after two years without any seedlings. It has been considered too early to take this step and we are considering a four year period of seedling absence instead.

Risks and impacts associated with intensively searching the islands

Our presence on the islands could be contributing to the weed problem. This was especially so during the early stages of control when the seedbank in the soil and the number of mature plants with setting seed was still high.

It is easy to prevent visitors bringing weeds onto the islands by following standard hygiene procedures prior to visitation, but it is very difficult to control the spread of seeds on staff footwear and clothing after weeding one site and moving onto another. It has been observed that many of the new sites encountered in the past two years have been on tracks or regularly-used pathways. The management regimes described in WESPKI are an attempt to reduce the amount of unnecessary travel and therefore the risk of spreading seed. Protocols of dusting seed off team members before leaving a site are followed but are not infallible.

Re-invasion from the mainland

Weeds have arrived on the islands in the past therefore re-introduction remains a threat. The main means of dispersal for these species is by wind and they may have got there by their own means. It is also possible that they arrived with earlier research parties, as they did not follow stringent island hygiene standards.

We do not know whether new sites are the result of new invasion or from existing seed in the soil being given optimum conditions to germinate. We do not know the extent of the seed shadow from the mainland, but the probability of seed dispersing 16 km and landing on an island in a location with suitable germination factors is low. All new Mexican devil, mistflower and mothplant sites have been in close proximity to existing sites, suggesting they have originated from the old sites. New pampas sites have occurred in remote areas away from old infestations. The pampas seed shadow may be more frequent than the other three weeds and the physical design of dispersal methods for the various seeds supports this theory.

The weed control programme and WESPKI model has been replicated on the Chicken Islands since 1997 and is showing promising results there. Hen Island has been included since 1998. However, we cannot logistically or financially complete control trips to all the islands over the critical pre-flowering period, and the Hen and Chickens programme is regarded as an intensive control programme. The intention is to contain and reduce infestations and allow natural regeneration to aid the process by means of displacement. We anticipate successful control will be achieved on these islands over a longer timeframe. It will be several years before we can confirm this, as some sites are yet to receive initial treatment. On the Hen and Chickens we have complete records of all sites, commencing with their first control visit, so the data will include earlier weed trends which are missing from the Poor Knights database.

Over five years our strategy has been refined with the knowledge gained from each visit. It is a case of successful techniques evolving and being developed by trialing ideas for management. On the Poor Knights Islands we have created a recipe that is successful at eradicating the five invasive weeds present and involves a balance between the timing of field trips, techniques for control and reducing interference and impacts of our visits. Attention to detail by those doing the work is paramount as to miss one plant prior to seeding has a significant bearing on the duration of the programme and whether the final result will be control or eradication.

ACKNOWLEDGMENTS

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Eradication planning for invasive alien animal species on islands – the approach developed by the New Zealand Department of Conservation

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Abstract New Zealand's Department of Conservation is now highly experienced in the field of invasive alien animal species eradication on islands, particularly rodent eradication. The approach which has been developed addresses eradication planning at an operational level and building capacity at an organisational level. At an operational level this is done by: planning the eradication operation to be as robust and as meticulous as possible to prevent the operation failing; avoiding failure of an operation which is frequently linked to a casual approach or a 'can't be done' attitude; recognising pre-conditions for eradication. These are: (i) all target animals can be put at risk by the eradication technique(s), (ii) target animals must be killed at a rate exceeding their rate of increase at all densities, (iii) immigration must be zero. Building of capacity has been achieved by: (1) strategic approach – planning island eradication programmes to maximise learning opportunities and minimise the risk of failure, (2) skills development - identifying training opportunities by participating in eradication operations elsewhere, (3) team approach - maintaining a committed project team and the support of higher level management, (4) peer review - an eradication advisory group provides advice on major pest eradication operations, (5) review and debrief – effectively transferring the lessons learnt with each operation to future projects. The approach outlined has application wherever eradication of invasive alien animal species on islands is planned.

DEVELOPMENT OF TECHNIQUES

NZ Department of Conservation and island eradications

Views have changed radically over recent years on the feasibility of eradicating invasive alien animal species from islands, as illustrated by historical rodent eradications. As recently as 1976, Yaldwyn (1978) concluded a conference on the ecology and control of rodents in New Zealand by stating that the possibility of complete extermination of rodent populations from New Zealand's offshore islands was "remote, or at least a very, very difficult thing indeed". By the early 1980s it was still widely held that no real breakthrough was in sight (Atkinson 1986).

New 'second generation' anticoagulant poisons became available in the 1980s. These poisons allow rats to consume a lethal dose well before they begin to experience poisoning symptoms (Taylor and Thomas 1989). This potency and late onset of toxic effects eliminated many of the causes of rats surviving poisoning using other pesticides. During the 1980s, using these poisons and a new eradication 'mindset', a number of rodent eradication operations were successful. Examples include the removal of Norway rats (*Rattus norvegicus*) from 9 ha Hawea Island in 1986 (Taylor and Thomas 1989) and from 170 ha Breaksea Island in 1988 (Taylor and Thomas 1993), plus the removal of Pacific rats/kiore (*Rattus exulans*) from 18 ha Korapuki Island in 1986 (McFadden and Towns 1991). Most early projects in New Zealand involved the removal of rodents from islands less than 200 ha in size by hand-laying poison baits.

By 1990 the practicality of eradicating rodents from small islands had been demonstrated and the possibility of removing rodents from islands greater than 200 ha in size began to be explored. Various eradication techniques were tested, such as a 'rolling front' of bait stations on Ulva Island (259 ha (Atkinson and Taylor 1992)) in 1992 (Thomas and Taylor 2002), and the aerial application of poison baits on Tiritiri Matangi Island, 220 ha, in 1993 (Veitch 2002b) and on Lady Alice Island, 120 ha, in 1994 (Ogilvie *et al.* 1997). By the late 1990s rodent eradications had been completed on two islands over 1300 ha in size – Kapiti Island off the south-west coast of the North Island and Whenua Hou/Codfish Island off the west coast of Stewart Island. The Department of Conservation progressively set aside funding for eradication projects on larger islands providing the impetus for a co-ordinated approach. A peer review team was established to provide advice on the allocation of funds and trials of eradication techniques.

The eradication of rats from Kapiti Island (1965 ha) was confirmed as successful in 1998 (Empson and Miskelly 1999) and Whenua Hou/Codfish Island (1396 ha (Atkinson and Taylor 1992)) was declared rat free in December 2000 (McClelland 2002). These operations represented new milestones in terms of island rodent eradication achievements. They were on islands eight times larger than any that rodents had previously been removed from in New Zealand and two species of rodents were eradicated on Kapiti Island. Until then, only single species eradications

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had been attempted in New Zealand. Both operations involved resolving non-target species issues. Each had to both achieve the eradication objective and to be another stage in a planned series of trials leading to the capacity to undertake even larger operations, such as the planned removal of rodents from Campbell Island (11,216 ha) in the remote Subantarctic (Atkinson and Taylor 1992).

CURRENT APPROACH

An Island Eradication Advisory Group has evolved from the peer review team and continues to focus on research, skills development, review and audit. The Advisory Group's efforts are currently directed at eradications on Tuhua/Mayor Island (1277 ha (Atkinson and Taylor 1992)), Raoul and Macauley (2938 and 306 ha (Atkinson and Taylor 1992)), Hauturu/Little Barrier Island (3083 ha (Atkinson and Taylor 1992)), and Campbell Island (11,216 ha). In early 2000 these were all in the early planning or implementation stage. All are more complex than the Kapiti and Whenua Hou/Codfish Island operations, and involve large islands, some with several species of invasive alien animals, and some in remote locations. That they are being attempted reflects confidence in the ability to plan and carry out successful operations.

One challenge facing the Department of Conservation, as eradication operations become more complex, is to ensure that effective communication and knowledge transfer take place within the organisation. It is vital that the lessons learned from each operation are recognised and disseminated.

The approach adopted addresses eradication planning at an operational level and capacity building at an organisational level.

KEY CONSIDERATIONS FOR PLANNING AN ERADICATION OPERATION

A number of issues must be dealt with in planning an eradication operation. Failure to consider any one of these can result in an unsatisfactory outcome.

The difference between eradication and control

Control operations manage the impacts of invasive alien animal species by sustained harvesting of the invasive species populations (i.e. reduced numbers of animals leads to reduced impacts). They are not concerned with removing the 'last animal'.

Eradication permanently removes the impacts of invasive alien animal species by eliminating the entire population. Pre-conditions for considering eradication are (Parkes 1993):

i. All animals can be put at risk by the eradication technique(s);
ii. Animals must be killed at a rate exceeding their rate of increase at all densities; and
iii. Immigration must be zero.

In planning an eradication the likely response of individual animals is important. Failure to recognise and account for individual variation in vulnerability could lead to survivors.

The following examples illustrate the importance of the behaviour and response of individual animals in an eradication operation:

i. Bitrex is added as a safety precaution to make some commercial rat bait less attractive to young children, by making it taste bitter. In laboratory efficacy tests involving bitrex in ICI rodenticidal formulations with albino rats and mice (20 animal groups, 3 day choice tests) some rats did not eat sufficient bait with bitrex in it to be killed (i.e. 3 out of 60 rats were not killed) (Kaukeinen and Buckle 1992). These tests are required as part of USA registration studies (EPA protocols). The ICI rodenticidal formulations all passed the minimum EPA test criteria of at least 90% kill, and led to the EPA's statement in their letter to ICI of 29 March 1990, that "The efficacy tests submitted for (bitrex-containing brodifacoum products) are acceptable" (Kaukeinen and Buckle 1992). The test result is acceptable for a control operation. However, for an eradication operation such information suggests a risk that should not be taken - it is not acceptable to have a percentage of the population not eating sufficient bait to be killed. On uninhabited islands closed to the public there is no benefit in including bitrex. Differences in consumption of bait containing bitrex and bait that did not contain it were observed in a 1996 trail on wild caught Pacific rats/ kiore on Hauturu/Little Barrier Island (Veitch 2002a).
ii. Over a four-year period 17 person-years of effort using traps, poison and hunting removed more than 3000 weka (*Gallirallus australis*) from Whenua Hou/Codfish Island. After all known weka had been accounted for (i.e. no sign could be found) 3 weka were located and caught using taped calls. These were all mature individuals who had been fully exposed to all previously used methods. Taped calls were used in an early phase of the weka eradication, but had not been used for over two years prior to their use in the last phase of the operation (Andy Cox pers. obs.).
iii. In the Kapiti Island eradication of brushtail possums (*Trichosurus vulpecula*) a total reliance on trapping as the eradication technique would have resulted in the operation failing. After a trapping effort of approximately 1,388,330 trap nights, dogs were used and the last 32 possums were found. Many of these animals showed signs of previous encounters with traps and may have been trap shy (Cowan 1992).

Eradication operations may use the same techniques as control operations, but the goal and therefore the essential mindset for everyone involved is different. If there is not the determination to remove every target animal and to plan, manage and implement an operation to achieve this goal, then there is a risk that the operation will be compromised. In addition the reasons for failure will be poorly understood. An eradication operation requires 100% focus and effort from all members of the project team.

Robust and meticulous planning

Planning for an eradication operation will involve research, contingency measures, incorporation of best available techniques and the flexibility to cope with unexpected difficulties. Biological, technical and logistical considerations such as seasonal variation in vulnerability of target and non-target species, type of bait and toxin used, and correct storage and presentation of bait are all taken into account.

Evidence (e.g. bait palatability, population dynamics and, non-target risks) supporting the assumption that a selected option will work must be critically reviewed. Techniques or operational practices that could expose the operation to an increased risk of failure need to be identified and avoided.

Data from previous control or trials on target population(s) must be considered. For example, if a proportion of the target population develops bait-shyness, toxin resistance or trap aversion due to pre-eradication activities, the eradication is likely to fail if similar techniques are deployed. Techniques used need to take into account past history. Toxic trials, if required, are usually carried out elsewhere to ensure that the eradication is not compromised.

On islands where new or complex scenarios are present, planning must ensure each targeted animal species is eliminated. Planning should be started early to identify issues to be solved. Trials are often essential to provide information or test modifications to existing methods. When developing new methodology test one thing at a time. Examples include:

i. The Tuhua Island eradication which is being used to trial methodology for planned concurrent rat and cat eradication on the larger and logistically more difficult Raoul Island. The trial includes clarifying the feasibility of eradicating cats by secondary poisoning and, if possible, determining what contingency technique would be appropriate for follow-up.
ii. Campbell Island's weather and large size make it impossible to use proven aerial bait application methodology without modifications. Non-toxic bait trials have been undertaken at Campbell Island to test the durability of different bait formulations and their acceptability to rats. Research has also been undertaken to define the most appropriate method for storage of bait in Subantarctic conditions, to ensure its optimum condition at the time of distribution by helicopter.

Timing of an operation is often critical. For example, poison baits will be delivered to any surviving cats on Raoul Island after their main prey, rodents, have been eliminated, but before they have substantial alternative prey available (i.e. nesting seabirds).

Identifying risks, and taking actions to eliminate or minimise them, is mandatory. If we are to learn from failed operations we have to identify possible causes of failure and act accordingly. For example:

i. In a rodent eradication operation, put poison bait on rock stacks around an island even if it seems unlikely that there are any rats on them.
ii. Test the toxicity of the bait before the operation to ensure it meets the minimum standards required for a successful outcome.
iii. Collect random bait samples during the operation for more detailed analysis in the event of a failure.
iv. Prior to the eradication operation take DNA samples from rats. If rats subsequently turn up it is possible to determine whether it was the eradication or quarantine precautions that failed, by comparing DNA samples with the pre-eradication samples.
v. Write operational standards and adhere to them. For example with a helicopter operation it is better to wait for suitable weather than fly in high winds and not achieve the necessary bait coverage.
vi. Assume that if something can go wrong it will and plan for it.

Successful eradication requires that all target animals are killed. To allow for variations in individual vulnerability due to age, behaviour, food supply, range size, etc., techniques must be 'over-engineered'. Therefore:

i. Lay more bait than you think you need.
ii. Despite a good shelf life use only fresh bait.
iii. Every trap must be perfectly set and sited. Each trap may be the one to catch the last target animal, or conversely the trap that loses and educates one of the remaining animals – so every trap counts.
iv. Re-sow even the smallest gap in bait coverage indicated on the navigational guidance printout, even though baits may actually be there due to the spread pattern using overlapping swaths which are conservatively set smaller than the sowing bucket actually delivers.
v. Take two helicopter buckets (one might break down).
vi. Use multiple eradication techniques for cat eradication (i.e. poison and traps).
vii. With cat eradication do not assume that no sign means no cats. Assess the probability of finding cats given the total level of effort that has gone into the eradication. A wide range of techniques (e.g. telemetry, trapping, searches using dogs etc.) and a lot of effort needs to go into eradicating cats. This is also important with other target species.

This attitude of 'over-engineering' should be adopted, not only by the project team but also by management. Managers often operate in an environment geared to cost efficiency. The focus for eradication must always be to eliminate failure. An eradication operation is more cost effective and more likely to succeed if it is carried out properly at the first attempt.

Overarching all the above is the need to rigorously monitor progress so that problems can be recognised and addressed.

Frequent causes of failure

Determination to succeed is essential in an eradication campaign. Where those involved take a 'casual approach', assumptions are not stated, questioned or tested. Scientific findings are often taken at face value without considering their validity in a new site or the relevance of that experimental design to other situations. For example, it cannot be assumed that bait stations work where more than one species of rodent is present. On Kapiti Island non-toxic bait trials revealed that Pacific rats/kiore would not use bait stations that Norway rats had used (Raewyn Empson pers. obs.).

A 'can do' attitude is essential, particularly in detecting and killing target animals at low densities. A project team has to be motivated and dedicated to achieve its goal. This requires considerable effort when few animals remain. For example, in the Kapiti Island possum eradication the last 32 possums were all located using dogs. This took 4502 man-dog hours (Cowan 1992).

Project teams must understand and agree with an eradication plan, know the importance of their role and how integral the effort of each and every one of them is to achieving a successful outcome. They also need to be aware of how they could compromise the operation by sloppy work. For example, poor servicing of bait stations could result in animals failing to get exposed to palatable bait and/or becoming bait-shy.

Eradication techniques (traps or toxic bait) must operate at optimal capacity to ensure a successful outcome. To achieve this the whole team must be motivated for the duration. Examples include:

i. In the latter stages of the Kapiti Island possum eradication dogs used to locate possums were periodically taken to the mainland to hunt where possums were common (Cowan 1992). This improved dog morale and handler confidence in the dog's ability to detect possums at extremely low densities.
ii. Difficult terrain combined with low to non-existent pest tallies in the latter stages of the Rangitoto Island possum and wallaby eradication proved a constant challenge to maintaining staff morale and motivation. This was met by:
    - using people with a positive attitude;
    - stimulating staff with other tasks (i.e. trips to other locations where they caught pest animals);
    - praise and acknowledgement throughout the latter stages of the eradication operation (Simon Mowbray pers. comm.).
iii. In the Whenua Hou/Codfish Island possum and weka eradication, motivation and morale was maintained by allowing the team to focus on the task of eradication. The team leader handled all other issues (e.g. resourcing, administration, and requests to do other work). Another essential element was involving all team members in the testing and development of better practice. This resulted in team 'ownership' of the techniques (Andy Cox pers. obs.).

A "can't do" attitude from other experts not involved in the operation can impinge on its success. Those planning an operation need to be explicit about the assumptions they are making and demonstrate that the planning has taken into account any points of concern raised by these experts. For example, the results of a study of possums on the West Coast of the South Island were used to justify the belief that the eradication of possums from Whenua Hou/Codfish Island was impossible and a recommendation was put forward to change the objective to control. The eradication project team believed that differences in habitat, climate and behaviour as a result of prolonged hunting pressure meant that the findings on the West Coast could not be used to predict the outcome. They were proved correct.

A "can't do" attitude by higher level management could have serious implications for resourcing, particularly in an extended programme. Operations need to be well justified and researched, robustly planned and documented, and effectively communicated with senior managers. Research, which measures the impacts and benefits of an operation, will help gain support for future operations and should be an integral component of all operations.

Building capacity at the organisational level

The New Zealand Department of Conservation has a commitment to learn from all eradication attempts, to reduce the risk of failed operations, and to build the capacity to attempt more complex projects. The approach adopted when planning invasive alien animal species eradication programmes on islands has several key components.

1. Strategic approach

By considering island eradication programmes collectively, rather than operation by operation, learning opportunities are maximised improving techniques for future eradication operations, and providing evidence of the benefits of eradication programmes.

2. Skills development

New project teams gain experience by participating in eradication operations elsewhere. This exposes team members to the reality of eradication operations and the issues and debate associated with them. This expands their horizons, builds up their network of contacts, and fosters the motivation needed to achieve a successful outcome. For example, the Raoul Island eradication project manager has been involved with planning aspects of the Tuhua cat and rat eradication, to maximise the potential for refining techniques that could be applied on Raoul.

3. A team approach

Major eradication operations require a committed project team. When appointing members, team dynamics must be taken into account. Project managers must be responsible for co-ordinating the respective contributions of team members to ensure programme goals are met within the agreed timeframe. It is important to have a well-briefed understudy for a project manager as insurance. Tasks need to be assigned explicitly to team members throughout the planning and operational phase. Team dynamics need to be considered, ensuring motivation and support are high and the skills required are transposed into clearly defined roles.

Project teams require the support of higher level management to effectively carry out their role. Pressures of other work often compromise time and quality of time spent on eradication projects. Time needs to be allocated and tailored to the requirements of an operation. For example a project manager may spend 25% of their time on the project in the initial planning phase, increasing up to 100% as approval to carry out the operation is being obtained, reducing back to 25% as contracts are let, and increasing up to 100% just prior to operation and throughout it.

Project teams must not operate in isolation. Each operation has local issues to address, but to ensure 'best practice' and skills development it is vital to involve people with relevant expertise and future project team members. This has the added bonus of sharing techniques and knowledge across a wider base.

4. Peer review

Peer review focuses on planning and readiness before an operation takes place. This is very important in operations that involve a 'single hit' technique. Everything has to be decided and all resources have to be assembled before a single animal is killed (i.e. in an aerial rodent eradication).

Peer review of major island eradication programmes now involves meetings between the Island Eradication Advisory Group, project managers, and other experts as required. Issues pertaining to current or upcoming island eradication projects are debated during these meetings. This has proved particularly beneficial to project managers, as it highlights points relevant to their project which may not have been raised.

The Island Eradication Advisory Group's brief is to provide expert advice to project teams and support managers in their decision to proceed with an operation. The group has been instrumental in getting organisational features (forward planning, skills, and review) operating across the Department. Focus is on:

i. Ensuring that lessons from past operations are transferred and that quality planning occurs;
ii. Looking ahead to the needs of future operations;
iii. Minimising political risks, which could affect the success of an operation or future operations;
iv. Ensuring island quarantine and monitoring is adequately planned for at the outset, to prevent re-invasion;
v. Continuing to act in an advisory capacity during eradication operations or when an alien animal species invasion is detected.

5. Review and debrief

Review needs to occur throughout all phases of an operation. Errors can be made during an operation and it can still be successful through sheer luck. Review assumes we cannot be lucky every time, and that mistakes should only occur once.

Debrief, at the end of an operation, assesses all aspects of the operation to determine possible improvements, make planning for future operations as robust as possible, and document successes. Debriefs effectively transfer the lessons learned with each operation to future projects, and involve current project team members and operators, as well as any contract staff, the Island Eradication Advisory Group, and project team members of upcoming eradication programmes. An example of this transfer of lessons learned relates to a failed eradication attempt. The documentation seen by the advisory group, before the operation, suggested planning for the project was adequate, but the eradication failed. Although some members of the project team had reservations about the project prior to the operation, these were not expressed until the operational debrief. This has led to an extra step in the planning process – members of the advisory group now visit the project team in the final stages of planning to check 'state of readiness' and does not rely solely on reports to the group.

THE FUTURE

Prevention

The next major challenge is improving the planning and implementation of island quarantine and contingency. Island quarantine consists of the precautions taken to minimise the risk of an alien animal species invasion. Contingency is the response to a new alien animal species invasion.

Island quarantine is particularly important as the number of successful eradication operations increases and we move into a situation where we are likely to be dealing with newly-established invasive alien animal species populations. Prevention is better than cure because it avoids the impacts of new invasive alien animal species establishing in vulnerable ecosystems.

Information dissemination

If we are to keep the eradication tools (i.e. toxins) currently available to us then we must use them wisely, and improve public understanding of the risks associated with using them and the benefits of successful eradication operations. To ensure that we all learn from island eradication attempts and to improve public understanding we need to make the results of eradication operations available through presentation and publication.

Wider issues

Although we have some understanding of the immediate benefits to threatened species of eradicating invasive alien animal species, we do not have a good understanding of the long-term effects of eradication, particularly the perturbations caused in an ecosystem by the removal of the alien species. Further work is required on defining long-term restoration goals for islands and island groups, so that invasive alien animal species eradication occurs within a context of restoration.

Refinement of techniques

There are further opportunities to improve eradication techniques and our understanding of how they operate. There is a requirement for more sensitive techniques for detecting and managing invasive alien animal species at low numbers, and for techniques which address issues associated with problematic animals such as those that have developed toxin resistance or become trap shy. Also, for more information and options for poison baits taking into account: bait life, palatability and attractiveness for a wide range of species.

The global challenge

The Department of Conservation approach has proved to work effectively in New Zealand where there are few native mammals and where invasive alien mammals are of special concern. We believe the aopproach has application elsewhere because invasive alien animals are a problem on many islands around the world. Many of the gains made in New Zealand have come about through forward planning, with each eradication supporting and leading on to the next. At the global level the challenge is to ensure that we all learn from all island eradication attempts. To do this will involve making the results of eradication operations available and developing effective international co-operation.

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Eradication of buffel grass (*Cenchrus ciliaris*) on Airlie Island, Pilbara Coast, Western Australia

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Abstract The aims of this project, now into the second year of the implementation phase, are to eradicate buffel grass (*Cenchrus ciliaris*) from the 25 ha Airlie Island off the Pilbara Coast, to develop and implement methods to restore the indigenous vegetation, and to collect and store seed for future restoration works. The most effective herbicides trialed under these conditions were Roundup Biactive 8 *l*/ha and Verdict 6 *l*/ha. Extensive field trials indicated the main perennial shrubby species on the island (*Acacia bivenosa*, *A. coriacea* and *Rhagodia preissii*) are resistant to Roundup Biactive and Verdict. With the exception of native grasses, these herbicides had no adverse effect on other indigenous plant species. Initial blanket and spot spraying with Roundup to kill the parent plants followed by blanket spraying, avoiding native grasses, with Verdict, is the most cost-effective regimen for control. A temporary (three years) water pipe for filling battery operated 250 *l* spraying units was installed across the centre of the buffel populations. Hoses 60 m long with hand held lances were used to apply herbicide. Four operators with two units can spray about two hectares each day. The best time for spraying is six weeks after heavy rain when the parent plants are actively growing and the new seedlings have grown sufficiently to spray. Spraying too early misses most of the seedlings; too late and the seedlings as well as the parent plants are seeding or too senesced to respond to the herbicide. The window of opportunity for spraying under these conditions is therefore only two weeks. Results of the spraying on Airlie Island indicate that 98% of the original stands of buffel grass has been controlled. Replanting with greenstock is preferable after heavy rainfall, the main shrubby species planted after spraying with Roundup can then be oversprayed, when required, with Verdict. Greenstock survival rates vary considerably between 5% and 90%, and are entirely dependent on follow-up rainfall. Two to four sprays a year, depending on rainfall events, are required for a period of at least three years (estimated age of soil seed bank) to control this weed with follow up monitoring and backpack spot spraying or hand removal. *Eulalia aurea*, a perennial dominant native grass, is best planted at the conclusion of the three year spraying programme to avoid spray damage and for ease of operations to control buffel grass.

Keywords buffel grass control; herbicide; restoration; marine.

INTRODUCTION

Buffel grass in Australia

Buffel grass (*Cenchrus ciliaris*) is a perennial grass native to Africa, the Middle East and southern Asia. It was first introduced to Australia in the packsaddles of Afghan camel drivers (Bryant 1962) and was later used by the pastoral industry for erosion control and as a pasture supplement throughout the Pilbara and Kimberley regions of Western Australia. This grass has also established on a number of islands off the Pilbara coast, including Airlie Island.

Introductions of buffel grass were primarily aimed at improving stockfeed, stabilising soil and revegetating bare and eroded areas (Bryant 1962; Humphrys 1974). The effectiveness of buffel grass at stabilising soils is due to the ready germination, rapid propagation and easy establishment, even on bare or infertile soils (Bryant 1962). Buffel grass is resistant to drought, fire and heavy grazing, so it is dominant and very persistent at a site once established making it useful as an arid zone pasture grass (Bryant 1962; Hodgkinson *et al.* 1989). These characteristics are attributed to the robust root system and swollen stem bases, which accumulate carbohydrate reserves, so that loss of leaf lamina during drought or after fire is not fatal to the plant. Regrowth may then be rapid in favourable conditions (Humphrys 1974).

Buffel grass favours alkaline soils (Christy and Moorby 1975; Griffin 1993). Within the arid zone it establishes best on areas of higher nutrients and moisture, especially creeklines and floodways.

There are several buffel grass varieties in Australia, each with different growth habits and requirements. Seed dormancy and germination characteristics may also be variable between or even within varieties. The varieties biloela, gayndah and others are African in origin and are widely favoured as pasture feed in Queensland. The Western Australian (WA) variety is shorter, reaching a maximum of 75 cm and not as vigorous (Humphrys 1974). Curiously, seedlings of the WA variety have lower drought tolerance, but flowers much more quickly following rains, and is hence able to survive and spread after rare rainfall events. The WA variety was originally introduced from the Middle East, and has possibly since become further adapted to arid Australian conditions.

In higher rainfall areas of Queensland, buffel grass does not spread rapidly, if at all, and usually requires cultivation to establish a population (Hacker and Ratcliff 1989). In the more arid conditions in central and western Australia, however, buffel grass (WA variety) is much more invasive. Its resistance to fire, drought and grazing make it extremely persistent, and its rapid growth and flowering allow it to dominate over native vegetation in some areas

Pages 92-101 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

(Humphries *et al*. 1991). The major mechanisms of dispersal are wind, flood, fire (Griffin 1993) and possibly domestic stock. Seed is also easily spread by humans as they readily adhere to trousers and socks, thus it is very important to make sure seeds are removed from clothing after visiting a buffel grass area. The spiked seed-bearing involucre also increases spread by attaching to animals. Vehicle wind-assisted spread along roads is also evident in Uluru National Park (Griffin 1993). Buffel grass was not reported as spreading until the 1970s, when high rainfall and floods lead to rapid colonisation along creeklines and alluvial flats (Griffin 1993).

The biology of buffel grass allows populations to be self-maintaining, and encourages rapid spread in favourable conditions. As a result, buffel grass (WA variety) is an aggressive coloniser of native habitats, especially moist environments, where it forms dense monocultures, excluding other species (Humphries *et al.* 1991). Buffel grass also alters fire regimes by increasing fire frequency and intensity, and while buffel survives, native species are suppressed or replaced (Griffin 1993; Humphries *et al.* 1991).

The success of buffel grass raises serious concerns for the welfare of plant and animal species that are restricted to moist sites throughout the arid zone (Humphries *et al*. 1991). These habitats are critical refugia for survival of numerous plants and animals (Griffin 1993; Humphries *et al.* 1991; ANCA 1996). Urgent control methods are required in central Australia (Humphries *et al*. 1991), especially in national parks and nature reserves.

Buffel grass has been reported as a weed or a serious invader in Western Australia in the following reserves and biologically significant areas: Cape Range National Park, western coastal plain of the Carnarvon-Exmouth area and Doole and Roberts Islands where it is potentially a threat. Numerous other islands in the Shark Bay - Exmouth area are exposed to invasion by buffel grass (Department of Environment, Sport and Territories 1996).

Buffel grass in Western Australia

In 1910 the first deliberate introduction of buffel grass in Western Australia was carried out on Wallal Station, Western Australia. Since then numerous introductions have been made, for example aerial sown seed on Mundabullangana Station between 1926-1928 (Bryant 1962).

Buffel grass seed was most likely brought to Airlie Island in soil used in the construction of the lighthouse in 1913 (WMC 1993). By 1987, buffel grass roughly occupied a 2.5 ha kidney-shaped area around the lighthouse (Astron 1988). Later that year Western Mining Corporation constructed an oil installation on the island, removing approximately 1.5 ha of buffel grass in the process (Astron 1988).

Following construction of the oil installation, buffel grass was estimated to be spreading at a rate of 0.2 ha per year and by 1993 had spread over 2.2 ha (WMC 1993). By the year 2000 it covered an estimated area of 8 ha.

The initial spread of buffel was relatively slow, however disturbance events appear to have enhanced its invasive capacity. The proposed decommissioning of the Airlie Island installation presents another disturbance event which could allow buffel infestation to further increase.

On Airlie Island the presence of buffel grass threatens the natural plant communities by replacing, almost entirely, the understorey cover of indigenous grasses and herbs. Buffel grass has already become a dominant plant species on the island and other islands along the Pilbara coast. This environmental weed substantially increases the fire risk which may impact upon the habitat of local fauna and may cause significant and permanent changes in vegetation structure and diversity.

At the start of spraying in 1999, buffel grass formed a near monospecific stand over eight hectares (33%) of the island. This weed may release allelopathic chemicals (Choo 1984) into the soil that inhibit growth of other species, potentially acting as a key displacement agent for most of the native vegetation. Buffel grass may also be detrimental to the island's fauna, especially the breeding cycle of shearwaters and the survival of herpetofauna.

Options for controlling buffel grass

A number of options were canvassed at the outset of the study including biological control which was deemed inappropriate because of potential adverse impact on the pastoral industry throughout northern Australia. Fire was ruled out as a control measure as buffel survives fire (Griffin 1993; Humphries 1991) and there is a complete fire ban on the island due to the risk associated with the oil storage facilities as well as impacts on the indigenous flora and fauna. Although isolated plants on the island are pulled up, physical removal is not usually appropriate due to the large number of plants and the difficulty of removal because of their strong root system. Other problems are the cost, soil disturbance and possible wind erosion (blow out from cyclonic wind). Mowing is ineffective, as well as costly, impractical (petrol mowers cannot be used due to fire risk) and possibly damaging to native fauna. The key option for research into the control of buffel grass focussed on herbicide control.

Key aims of the project

Phase one of this project was to investigate and research the biology of buffel grass and to develop a control programme which will integrate eradication or sustainable control of buffel grass with the reinstatement of indigenous species.

Phase two of the project is the control of buffel grass over the whole island based on the results of Phase 1, restoration of indigenous communities, and initiation of a seed collection and storage programme for future revegetation works. This phase has been underway for two years, and is the emphasis of this paper.

Outcomes of this study will be directly relevant to the decommissioning of the Airlie Island oil installation while providing benchmark data on the control of buffel grass. The study will therefore be of regional, national and international significance to land managers and conservation agencies where buffel grass is an environmental weed. Information on buffel grass control generated from this programme is already being utilised by local land managers on adjacent islands and adjacent mainland as well as other areas in Australia such as Queensland and central Australia.

Data was generated in Phase one of the project including buffel grass seed production/viability; longevity of the soil seed bank (at least three years); buffel spread; densification and percentage groundcover; seedling recruitment; soil nutrient profiles; indigenous plant resistance to herbicides (8 *l*/ha Roundup no effect on *Acacia bivenosa*, *A coriacea* or *Rhagodia*); life cycle of buffel (seed can germinate and flower in six weeks, plants can grow, germinate, flower or seed any time of the year as long as conditions are favourable, a minimum of 10mm rainfall) and comparison with *Eulalia aurea* the dominant indigenous grass species. Phase one also found that repeated spray trials (Roundup then Verdict) after regrowth gave excellent kill rates as did seedling spray trials. Detailed summaries of the results of research undertaken in Phase one are being prepared for publication.

Study site

Airlie Island (Fig. 1) is a 25 ha nature reserve and lies 35 km north-east of Onslow.

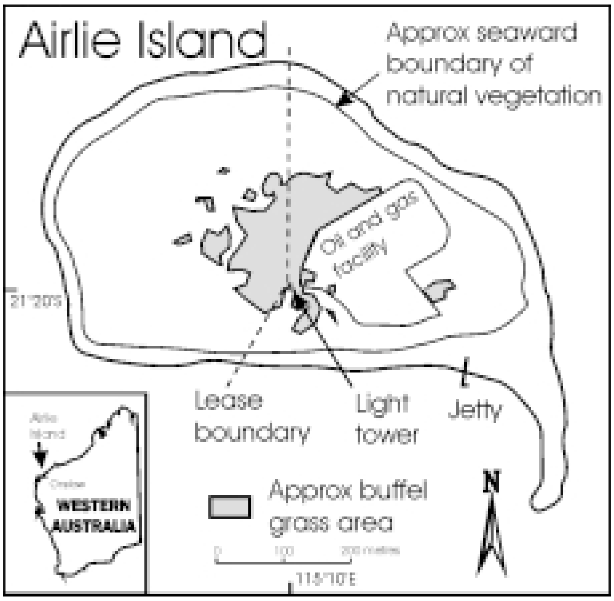

Fig. 1 Airlie Island with infrastructure, lease boundary and approximate buffel grass area when eradication work began.

The island has been used in the last decade as a fuel storage depot for Western Mining Corporation Petroleum Division (WMC), Novus Petroleum's offshore oilfield projects and by the present lease holder Apache Energy. Imminent decommissioning of the petroleum activities on the island require that buffel grass control be undertaken to ensure that the weed is contained, controlled or eradicated. Though the lessee is only required under agreement to control buffel grass and revegetate in the leased area (approximately half of the island), they are undertaking a programme to try to eradicate buffel grass from the entire island and revegetate areas, where necessary.

Airlie Island is on the borders of the Fortescue and Carnarvon Botanical districts (WMC 1988), and receives an annual rainfall of 300 mm from both summer (cyclonic) and winter (cold front) storm systems. Much of this rain falls intensively in irregular intervals, often accompanied by severe winds. The soil on the island is white to pinkish, generally coarse, calcareous sand with poor water holding capacity.

The rainfall pattern, lack of permanent surface water and small size of the island means the vegetation must be drought tolerant and able to cope with salt-laden wind. Most of the island is dominated by two *Acacia* species (*A. bivenosa* and *A. coriacea*), with *Rhagodia preissii*, *Eulalia aurea* and now buffel grass (*Cenchrus ciliaris*) as the major perennials. Shorelines are dominated by *Spinifex longifolius*, *Ipomoea pes-caprae*, *Sporobolus virginicus* and *Eulalia aurea*. During favourable seasons, a large number of annual species can be found in abundance, including *Portulaca intraterranea*, *Euphorbia* spp., *Boerhavia repleta*, *Cleome viscosa*, *Cuscuta australis* and *Threlkeldia diffusa* (WMC 1988).

Only two other weed species are known to exist on Airlie Island. Cotton bush (*Aerva javanica*) plants may be found anywhere on the island. When found during spray operations these are pulled up and any fallen seed is picked off the ground; plants are placed in sealed polythene and taken off the island with all other rubbish for disposal (deep buried). Any plants seen by Apache Energy staff are treated in the same manner. Only a few plants are found each year; locations of these plants are noted and occasionally these sites are visited by Apache staff to check for new plants. The other weed is a native, *Abutilon lepidium*, that occurs naturally on adjacent islands but not on Airlie. To date about 300 seedlings have been removed from one small area (Long pers. comm.).

METHODS

Aerial Photography and Image Enhancement

Aerial photographs of Airlie Island are taken annually. Visual examination of these images from 1993 to 1998 shows only a few small clumps of buffel grass colonising away from the main buffel zone. There are numerous small plants known to exist that do not show up on the photos. The major feature of the series of images is densification and gap-filling within the existing stand, which is responsible for most of the increase in buffel grass cover on the island.

The set of photographs taken in June 1997 were scanned and colour enhanced for the wavelengths corresponding to buffel grass, beach spinifex and *Eulalia*. These areas were coloured yellow, purple and green respectively to enhance visualisation of the extent of buffel grass spread.

Selection of herbicides

The key aim of the research programme was to identify a safe and effective herbicide to control buffel grass with minimal impact on the indigenous flora and fauna. Buffel grass was known to be very difficult to control due to its ability to survive long periods of drought and resprout from dormant buds. Other species with similar traits (e.g., *Ehrharta calycina*, *Eragrostis curvula* and *Hyparrhenia hirta*) can be successfully controlled using the correct herbicides and spraying at the right stage of growth. For example, spraying *Ehrharta calycina* before flowering results in a high death rate, whereas spraying at flowering time results in dieback of the top part of the plant but resprouting from dormant buds at the base (Dixon 1999). Herbicides known to control other *Cenchrus* species (e.g. *Cenchrus echinatus*), and which would be suitable for Airlie Island conditions, are the grass-selective herbicides Fusilade 212 and Verdict 104 (Parsons 1995).

Successful trials and large-scale herbicide application on other grasses have been conducted in Kings Park bushland, Perth, Western Australia, indicating several of the grass-selective herbicides are safe to use over most indigenous plants including monocotyledons such as sedges and rushes, kangaroo paws (*Haemoderaceae*) and orchids, and dicotyledons such as the Proteaceae family. As we had extensive experience with the following herbicides, the first trials included the grass selective herbicides Fusilade 212® (212g/*l* fluazifop-p) Fusilade WG® (212 g/kg fluazifop-p) (this formulation of wettable granules is easier and safer to handle and does not smell of hydrocarbons), Targa® (99.5 g/*l* quizalofop-p-ethyl) and the non-selective herbicide Roundup Biactive® (360 g/*l* glyphosate) (the wetting agent in this formulation is claimed to be safer to use over fauna, particularly frogs). It should be noted Roundup® 360 (360 g/*l* glyphosate) and other glyphosate formulations had been tried by several people to control buffel grass in central and eastern Australia, their results were very poor and we were advised not to use it. Verdict® 104 (104 g/*l* haloxyfop) was included in the latter trials. These herbicides were evaluated over a three year period, not all the trials will be included in this paper.

Redeye®, a marker dye incorporated in the spraying tank, was used in the larger trials to make sure there was no area missed and over-spraying was avoided. Though Redeye is a Schedule 6 poison (Australia-wide schedule outlining the toxicity of pesticides) and widely available to the general public, there was no other formulation available, to our knowledge, that would remain evident for several days under the hot spraying conditions experienced on Airlie. We also decided not to use any spraying oils for fear of damaging foliage in the high temperatures and thus inhibiting the translocation of herbicide into the buffel plants.

Spray plots

Spray plot sizes were 1x1 m, each separated by a 0.5 m-wide corridor. Three replicas of each plot were made within each trial site. All plots were sprayed working systematically across the plot and back again in the opposite direction to ensure even coverage. Compressed sheeting was used as a barrier to ensure there was no drift onto other plots.

Trial I

Condition of plants before spraying was scored using Table 5 based on the experience of the authors.

Sprayed in February 1997 when the plants were showing vigorous new growth after the summer wet season, but showed some signs of drying out (spraying condition 4 going on 6).

Treatments used were: Fusilade 212, 2 *l*/ha (plus Agral 60 @ 1 ml/*l*); Fusilade 212, 4 *l*/ha (plus Agral 60 @ 1 ml/*l*); Fusilade WG, 5g/*l* @ 2 *l*/ha (plus Agral 60 @ 1 ml/*l*); Fusilade WG, 10g/*l* @ 4 *l*/ha (plus Agral 60 @ 1 ml/*l*); Roundup Biactive, 3 *l*/ha; Roundup Biactive, 6 *l*/ha; Control.

Trial II

Sprayed in October 1997 when the weather was warm and the buffel grass showing signs of drought stress. The buffel grass was drying out, and had only a few obvious green shoots (Spraying condition 6, Table 5).

Treatments used were: Roundup Biactive 0.6,1.5 and 3 *l*/ha; Targa 6 and 8 *l*/ha + Agral 60 @ 1 ml/*l*; Verdict 6 and 8 *l*/ha + Agral 60 @ 1 ml/*l*; Fusilade 212, 6 and 8 *l*/ha + Agral 60 @ 1 ml/*l*; Control.

Trial III

Sprayed in April 1998 in low to moderate wind conditions. Plants were responding to recent heavy rain with vigorous new growth, in spraying condition 3 (Table 5). This first rain was followed by regular rainfall events for the following 11 weeks.

Treatments used were: Fusilade 212, 6 and 8 *l*/ha + Agral 60 @ 3 ml/*l*; Targa 6 and 8 *l*/ha + Agral 60 @ 3 ml/*l*; Verdict 6 and 8 *l*/ha + Agral 60 @ 3 ml/*l*; Roundup Biactive, 0.6, 1.5 and 3 *l*/ha; Control.
NOTE: The rate of Agral 60 was altered from the previous two trials (3 ml/*l* up from 1 ml/*l*) following the manufacturer's recommendations.

Large-scale spraying trials

A few large-scale ($100m^2$) trials were attempted with herbicides achieving a high kill-rate in the multi-herbicide trials. These trials using a 15 litre back-pack sprayer were aimed at confirming the effectiveness of herbicides sprayed on a larger scale prior to the implementation phase.

Roundup Trials

Three large areas were sprayed in April 1998 with 6 *l*/ha Roundup Biactive as a broadscale trial. Plants were in spraying condition 3 (Table 5).

Further trials were sprayed at 8 *l*/ha under spraying condition 4 (Table 5).

Verdict Trial

A large area around the buildings was sprayed in July 1998 with Verdict at 8 *l*/ha. Plants were in spraying condition 4 (Table 5).

Implementation phase

This phase involved spraying all of the buffel grass on the island with approved, effective equipment. Some of the key constraints which we had to address are as follows:

- Getting the equipment on the island. This was trucked up to the remote townsite of Onslow then placed on a barge which goes to the island once a week. No large, heavy or dangerous items can be carried on the helicopters which take personnel to the island from Barrow Island, the direct flight destination from Perth.
- Using the best equipment for the job and making sure we had adequate spare parts. No internal combustion engines are allowed on the island (due to possible hydrocarbon contamination/spills), therefore we had to use battery-operated spraying units. Water supply for spraying is from a reverse osmosis supply generated on the island, rainwater also goes into the tanks. This supply is limited; on one occasion the pipe to the tanks burst and we only had just enough water for spraying operations.
- Cyclones during spraying operations. When these develop the island is evacuated and all operations ceased.
- Laying a temporary (three years) water pipe with taps for filling the spray tanks.
- The 250 litre spray tanks have to be carried to each station by hand, no vehicles allowed in the natural vegetation areas. It is therefore necessary to judge very carefully how much material you need in the tank to finish off the area.
- Coping with the extreme weather conditions (e.g. 36°C and 80% humidity during summer spraying operations) and difficult working conditions (e.g. walking backwards when spraying) and abundant dead twigs which fouled boots and clothing.
- Training for the job and safety issues and the need for annual renewal of Helicopter Underwater Escape Training (HUET).

Equipment used

2 x 250 litre sprayers with 12 volt battery operated pump; 2 x 60 metre reel hoses and adjustable spray guns; 1 x 12 volt battery charger; 6 x 12 volt car batteries; 600 metres x 32mm ID pipeline; 10 taps miscellaneous joiners/fittings; 2 Hardie backpack sprayers.

Full-scale spraying operation

Roundup Biactive at 8 *l*/ha and Verdict at 6 *l*/ha were used. The strategy employed Roundup in the first spraying operation as there were few annual indigenous plants emerged that might be affected by the spray. Then Verdict used as a blanket spray as annuals and possibly some native perennial plants are at a susceptible stage.

Each spraying unit requires two operators, one as a sprayer and the other to release and withdraw the hose. Operators needed to change duties on a regular basis to avoid fatigue as a result of the high temperatures. Operators avoid spray drift by spraying upwind which can be difficult as you have to spray around bushes and under them whilst avoiding spray application of the bushes. In dense stands we use a blanket spray and other areas spot spray. Spray is applied in a circular fashion, pulling the hose out and spraying as you go to the extremities of the hose, then working backwards to the spray unit and again pulling the hose out until the entire circle is completed. The unit is then moved onto the next section. Some areas not covered have an extra section of hose added to cover the area; alternatively we use a backpack sprayer for outliers or remove isolated plants by hand. Most of the leased area of the island can be accessed by using a road or around the bund surrounding the oil tanks. In these areas we placed the spraying tank on a tractor for ease of movement.

Round One Spraying - March 1999

The first round used Roundup for ease of operations and because it is more cost effective than using Verdict. Herbicide application was following six weeks of heavy soaking rain. Ideally the spraying programme should have been initiated two weeks earlier as most of the buffel grass had just reached first anthesis and early seeding. The buffel grass was in rapid growth with some plants beginning to dry out by the fourth day of spraying. Most of the application was blanket spraying of heavily-infested buffel areas, avoiding as many indigenous plants as possible with very few annual seedlings present. Previously-sprayed areas were spot sprayed, avoiding contact of Verdict on *Eulalia* to reduce damage to planted greenstock and annual indigenous plants. Though original trials indicated *Eulalia aurea* was resistant to the grass-selective herbicides, probably because they were under stress at the time, subsequent trials showed they were very sensitive and future spraying would need to avoid excessive contact with *Eulalia*.

Round Two Spraying - June 1999

The second round of spraying was mainly blanket spraying resprouts and seedlings with Roundup, as there was such a large amount to spray and low levels of annual indigenous plants in the previously-sprayed area. Some spot spraying with Roundup in outlier areas and spot spraying Verdict over replanted areas was done.

Round Three Spraying - April 2000
The third round of spraying followed six weeks of substantial rainfall on the island as a result of a cyclone. The buffel was in excellent condition for spraying. About half of the area, low impact areas, were sprayed with Roundup; the remainder, rich herb fields, with Verdict. Backpack sprayers were used on outlier populations. A thorough inspection after the spraying operation detected some other plants which were removed by hand.

Revegetation

Greenstock propagation
All seed was collected from Airlie Island to ensure only local provenances were utilised. Plants of *A. coriacea, A. bivenosa, Rhagodia preissii* and *Eulalia aurea* were propagated from seed in glasshouses at Kings Park and Botanic Garden and transported to Airlie Island. *Acacia* seed were hot-water treated and left to soak overnight. Seed was sown into punnets and after about six weeks, seedlings were pricked out into tubes. Seed was sown in summer for planting in early winter of the following year. Propagation of larger seedling numbers was in deeper 5cm x 5cm x12cm pots to promote stronger root development and to give a longer holding period in case of dry conditions. In 1999 plants were raised in an 'Accredited' (approved by the Nursery Industry Association of Western Australia) commercial nursery, which was inspected beforehand and during the growing period to ensure adequate hygiene and weed-free propagation. Using an accredited nursery as the source of all plant material reduces the risk of introducing pests, diseases and other environmental weeds.

Transport of the seedlings to the site was by truck to Onslow, then barge to Airlie Island. Plants were watered prior to packing and transport to the trucking company. Polystyrene foam boxes and strong waxed cardboard boxes were used to transport the seedlings.

Replanting density
Benchmarking (using quadrats and transects) in pristine areas of the island gave a figure of natural plant density and the estimated number of plants required for revegetation of bare areas. This density was doubled to allow for assumed seedling death. The total area requiring revegetation is estimated at one hectare; this excludes the leased area which is to be revegetated after decommissioning the oil facilities.

Planting Times
The lack of summer rain resulted in plantings only being undertaken in winter.

As soon as the plants arrived on the island, they were placed in a shady position and watered. Planting was done with a garden trowel, the root balls of the plants were placed just below the soil surface to avoid drying out. The first planting trial in April 1998 consisted of *A. bivenosa, A. coriacea, Rhagodia preissii* and *Eulalia aurea* which were planted into moist soil. A further trial in June/July used only 200 *A. bivenosa*. A large trial in June 1999 used 1400 plants: *A. bivenosa* and *E. aurea* in high numbers; *A. coriacea* and *R. preissii* in low numbers. These were planted in very dry conditions. Planting was in a number of locations in areas previously heavily-infested with buffel grass. After cleaning with bleach and thoroughly rinsing, the spraying units and long hoses were utilised to water the plants in.

RESULTS

Aerial Photography and Image Enhancement

Aerial photographs taken in June 1997 were scanned and colour enhanced for the wavelengths corresponding to buffel grass, beach spinifex and *Eulalia aurea*. From this it was estimated that 6 ha, or 23% of the island was infested with buffel, and that there was virtually no overlap in the distribution of buffel grass and *Eulalia aurea*. Further data collected in 2000 indicated that buffel grass actually covered almost 8 ha of the island.

Spray Plots

Trial I

The trial was scored 11 weeks after spraying (Table 1) and assessed again after six months. This later assessment, after some resprouting from dormant buds, found no dead

Table 1 Spraying Trial I; applied in February 1997.

Herbicide	Concentration and application rate	Percentage live plants flowering	Percentage death of above ground biomass
Roundup Biactive	7.5ml/*l* @ 3 *l*/ha	<40%	88% ± 7%
	15ml/*l* @ 6 *l*/ha	55%	88%± 7%
Fusilade 212	5ml/*l* @ 2 *l*/ha	65%	4%± 4%
	10ml/*l* @ 4 *l*/ha	68%	22%± 12%
Fusilade (WG)	5g/*l* @ 2 *l*/ha	77%	0 %
	10g/*l* @ 4 *l*/ha	53%	23%± 20%
Control	—	100%	0 %

plants in the control or any Fusilade plots (except one liquid application @ 2 l/ha in one plot, in which 20% of the plants had died). In contrast, most plants were dead in all Roundup trials. Plants that were still alive were flowering. A similar situation was found 18 months after spraying, but numerous germinants had emerged in the Roundup plots.

A comparison of seeds collected 11 weeks after the trial from sprayed and unsprayed buffel showed that all the florets of sprayed plants were empty, compared with the unsprayed plants, which had 80% of florets with seed, with an average of 1.50 ± 0.18 seeds per floret. It appears that although spraying under non-optimal conditions is not very effective at killing plants, it may prevent viable seed-set, and therefore be worthwhile.

Conclusion

Both Fusilade formulations had low kill rates (~20%) at the higher concentration of each. Roundup had a better kill-rate of (~80%), with most plants succumbing the following drought period. The dry weather conditions at the time of spraying limited the effectiveness of the herbicides, although Roundup had achieved complete kill when it was reassessed in August.

Trial II

The trial was scored six months after spraying, when the next rains fell.

The trial was scored again in July 1998, nine months after spraying. All plants were alive, resprouting from dormant buds.

Conclusion

No herbicides caused any buffel plants to die, owing to the unfavourable spraying conditions. These results show the importance of spraying plants at the correct stage of growth, as it appears that stressed plants were drying out and resulting in a lack of translocation of the herbicide from the leaves through to the dormant buds at the base of the plant.

Table 2 Effectiveness of herbicides on kill of buffel grass in trial 2.

Herbicide	Concentration and application rate	Percentage death of above-ground biomass
Roundup	1.5ml/l @ 0.6 l/ha	14% ± 8 %
	3.75ml/l @ 1.5 l/ha	0 %
	7.5ml/l @ 3 l/ha	14% ± 14 %
Fusilade 212	15ml/l @ 6 l/ha	20 %± 20 %
	20ml/l @ 8 l/ha	0 %
Verdict	15ml/l @ 6 l/ha	94% ± 6 %
	20ml/l @ 8 l/ha	0 5%
Targa	15ml/l @ 6 l/ha	12% ± 6 %
	20ml/l @ 8 l/ha	11% ± 11 %
Control	-	0 %

Table 3 Effectiveness of herbicides on kill of buffel grass in trial 3.

Herbicide	Concentration and application rate	Percentage death of above-ground biomass
Roundup	1.5ml/*l* @ 0.6 *l*/ha	57% ± 4%
	3.75ml/*l* @ 1.5 *l*/ha	52% ± 10%
	7.5ml/*l* @ 3 *l*/ha	67% ± 5%
Fusilade 212	15ml/*l* @ 6 *l*/ha	71% ± 18%
	20ml/*l* @ 8 *l*/ha	100% ± 0%
Verdict	15ml/*l* @ 6 *l*/ha	95% ± 5%
	20ml/*l* @ 8 *l*/ha	95% ± 5%
Targa	15ml/*l* @ 6 *l*/ha	44% ± 5%
	20ml/*l* @ 8 *l*/ha	34% ± 23%
Control	-	0%

Trial III

The trial was scored in July 1998, 10 weeks after spraying. During this time rains had been consistent, with several large falls. Conditions for plant regrowth were exceptionally good, so the results in Table 3 are the worst (i.e. maximum regrowth) that could be expected from a spray applied in good conditions.

These results suggest that, under optimal spraying conditions, Targa is of limited use, Roundup provides moderate kill-rates, and Fusilade and Verdict both give excellent results, killing nearly all plants (above-ground biomass) sprayed. Examination of larger Roundup trials showed that plants resprouted from the spray-shadowed portion, killing only leaves that were sprayed directly. Verdict and Fusilade, however, seemed to usually kill the entire clump (above-ground biomass) in a single application. Verdict worked equally well at both concentrations, while Fusilade appeared to work better at the higher concentration, which killed (above-ground biomass) all plants sprayed. Some resprouting of all treatments occurred at a later date but not as much as usual, indicating the plants were sprayed under ideal conditions.

Condition of Plants

The three multi-herbicide trials described previously were applied under different conditions, and showed a marked variation in effectiveness. The major difference between trial conditions was the condition of the plants. The spraying conditions and results are listed in Table 4.

Plants that were vigorously growing were killed more effectively, while senescing plants were not killed, or in the case of Roundup, killed up to the onset of leaf desiccation.

Spraying conditions

Buffel grass must be actively growing (in spraying condition 3 to 5, Table 5). Do not spray senescing plants.

Table 4 Susceptibility of buffel grass to herbicides at different stages of plant condition

Plant Condition*	Herbicides achieving <35% kill	Herbicides achieving 35%-75% kill	Herbicides achieving 80%-100% kill
3. Vigorous new growth	Targa	Roundup, Fusilade at lower concentration	Verdict, Fusilade at higher concentration
4. Vigorous growth and flowering	Targa	Targa, Fusilade at lower concentration, Roundup at all concentrations	Verdict all concentrations, Fusilade at higher concentration
6. Senescing	Targa, Verdict, Fusilade most conditions	Roundup (if plants just starting to senesce)	No herbicides achieved this kill rate

* Full description of plant condition in Table 5

Large scale spraying trials

Roundup

All areas sprayed at 6 *l*/ha had a high rate of kill where Roundup had been directly applied to leaves, but plants and portions of plants within the spray shadow were resprouting. It is therefore necessary to spray a second time when the buffel has had time to resprout.

The results from the 8 *l*/ha trials were outstanding with very few resprouts indicating this is the appropriate rate for use for the implementation programme.

Verdict Trial

Verdict at 8 *l*/ha, sprayed around the buildings in July 1998 gave excellent results, killing all the seedlings sprayed and most of the parent plants.

Conclusion

Roundup has a very high kill-rate where it can be applied directly to the leaves, but for large plants a second spray is needed after the above ground biomass death and subsequent resprouting to destroy plants and culms which were in the spray shadow. Verdict is ideal for spraying seedlings and will also kill previously-sprayed resprouting parent plants as well as some plants not previously sprayed.

Implementation phase – the eradication of buffel grass on Airlie Island

Round One Spraying - March 1999

Most of the buffel could be seen to be yellowing by the fourth day of spraying, indicating a good kill rate. Coverage appeared complete, but occasional unaffected shoots were detected as had been previously observed in herbicide trials.

Cyclonic rain shortly after spraying led to an unexpected rapid regrowth of large plants, however conditions by the end of April were too dry to spray. Desiccation of plants and the combined effect of the spray prevented seed set on the resprouting culms.

Some plants missed spray application, as expected, mainly under acacias as well as occasional plants away from the main populations. One area which was showing signs of drying out towards the end of the spraying period exhibited reduced death rates. However, overall results were better than anticipated with an estimated 80%-90% kill rate of mature plants, though there was resprouting on a number of plants.

Table 5 Buffel grass plant condition and suitability for spraying

Condition Number	Plant Growth Stage	Suitability for spraying
1	Fully dormant (no visible green tissue).	Plants not growing. DO NOT SPRAY.
2	Bud-break (no extended leaf lamina yet visible).	Not enough leaf area for herbicide uptake. DO NOT SPRAY.
3	At least half of shoots with extended leaf lamina. Not yet flowering.	Conditions acceptable for spraying.
4	All shoots with extended lamina. Flowering evident.	Conditions perfect for spraying.
5	Vigorous shoot growth. All mature culms fully extended.	Conditions perfect for spraying.
6	Any evidence of senescence of leaves or culms, i.e. leaf-rolling or desiccation (leaf margins dry).	Plants not growing. DO NOT SPRAY.

The method of spraying, though time consuming, worked well. Spray equipment was adequate and effective with no breakdown, and the batteries lasted longer than predicted (five hours) before recharging was necessary. The correct positioning of the water pipe and taps assisted smooth operation, and judging the appropriate amount of spray to finish spraying an area worked well. On average, four operators with two units can spray two hectares each day, or four days to spray the entire buffel-infested area.

Round Two Spraying - June 1999

As the parent plants had decreased significantly in number, regrowth was easier to target with spray. There was a significant increase in the number of seedlings, but not as high as anticipated in previous trials; this may be due to the drier conditions experienced in 1999. The extent of the buffel infestation appeared not to have increased or decreased, but the area took less time to cover as most of the main stand of buffel had been decimated. The kill rates were high, eliminating most of the resprouting plants and other parent plants that were missed in the first spray. There was little damage to non target species and certainly no recorded damage to the perennial species, with the exception of some damage to young *Eulalia aurea* plants in revegetation trials where Verdict was sprayed.

Round Three Spraying - April 2000

The third spraying was mainly of seedlings, as few mature live plants remained. At this stage we estimate most of the seedlings and over 98% of mature buffel grass plants on the island were eradicated. Seedlings may germinate from the soil seed bank. However, the viability of the soil seed bank is rapidly declining and sustainable control of buffel grass will depend upon careful monitoring and a judicious ‘mop-up’ spray programme.

Revegetation

Greenstock propagation

The seedlings produced both in the small trials and by the accredited nursery were healthy and vigorous, and no weeds were present in the pots. The most successful container for transportation was found to be polystyrene foam boxes as strong waxed cardboard boxes became moist and collapsed, causing some damage to seedlings.

Replanting density

The density of planting in our trials proved successful and we recommend a planting rate that would result in *A. bivenosa* and *A. coriacea* at a density of 1250/ha and *Rhagodia preissii* and *Eulalia aurea* at a density of 2500/ha.

We recommend that planting densities be double to allow for a 50% death rate. However, we must be vigilant as *A. coriacea* has declined in some areas and it is much slower growing when compared to *A. bivenosa* which may eventually outcompete it and require thinning.

Planting

The results from the first two planting trials were outstanding with 89% to 97% of all species surviving and growing vigorously 10 weeks after planting.

Further monitoring in April 2000 indicated there was no further death in the April or June/July 1998 trials. This may have been due to substantial rainfall after planting.

The large planting in June 1999 was a failure with no rain for six months after planting. Almost all the plants of all species died. Though no exact count was done, a few plants of *A. bivenosa* are alive in two or three locations, with survival of <1%.

DISCUSSION

The research results from Phase one of the programme underpinned the implementation phase of the programme. A good knowledge of the biology of the buffel plants opened up a narrow window of opportunity for spraying and optimising kill rates, and seed production, soil seed bank and longevity of seed enabled us to plan ahead with a work schedule. This is aimed at eradication of buffel grass from Airlie Island in the near future, as long as the appropriate funding is available.

Spraying needs to occur whenever there is sufficient rainfall to control the few remaining mature plants and the emergent seedlings of buffel grass. As the estimated age of the soil seed bank for buffel grass (Phase 1 unpublished data) is three years, rapidly declining after eighteen months, further large-scale spraying operations will not be necessary. Trials with motorised, battery operated backpack sprayers indicate this is the best option for spraying smaller numbers of plants. Hand-operated units are too debilitating for operators in the hot, humid conditions experienced on the island.

The initial trials provided an appreciation of the problems and how adaptable you have to be with your implementation programme which is entirely dependent on the vagaries of the weather. The spraying results proved to be inconsistent as every trial was different. However, the results from the large-scale spraying programme were used as the basis for the implementation phase. The authors are confident that the herbicides, and their concentrations, used in the implementation phase are ideal for the expected varying conditions of the plants. Future operations, due to the recruitment of herbs in the buffel areas, will be using Verdict to avoid damage to these plants. Planting *Eulalia aurea* should be delayed until the soil seed bank is eliminated, otherwise further spraying could kill or damage *Eulalia.*

The best time for spraying adult plants is within the range of three to five weeks after sufficient rain has fallen. If spray is applied too early the seedlings are too small to target and with extra rainfall there may be delayed germination. Also it is not cost-effective to spray twice when one operation can achieve good results. Under quick drying conditions some plants are beginning to get stressed, and may occasionally drop some mature seed before they are sprayed.

Logistically there are major problems with a remote insular site such as Airlie Island. For example we have to rearrange our usual work schedules when the spraying is necessary as rainfall in the arid zone is erratic. All the accommodation, flights and so on are arranged by the oil company and, if urgent work is being done on the oil installations, securing necessary flights and accommodation can be problematic.

Though the spraying programme is in hand, the programme has identified an urgent need to revegetate bare areas. The dead below-ground biomass of buffel does provide soil-binding to prevent wind erosion for at least three years, in which time revegetation needs to proceed.

Planting greenstock can give excellent results if the ground is moist and there is follow-up rain. We recommend the best time for planting is in late autumn or early winter, though it can also be dry at this time of the year as experienced in the 1999 trial.

High on the priority list is a comprehensive seed collecting and storage programme of all species native to the island, with key areas being the rich herbfields between the Acacias and other perennial plants when decommissioning takes place. Some stabilisation trials also need to be undertaken as cyclonic winds are a regular occurrence during the summer season. A recent cyclone altered the shape of the island and caused erosion near a flare installation which is being re-stabilised and revegetated by consultants based on the mainland.

The ongoing success of the control of buffel grass and revegetation of Airlie Island with indigenous species depends upon the good aegis of the funding sponsors who manage the island. The programme does show that with careful, focussed research, it is possible to achieve effective and timely weed control in the arid zone.

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Eradications of invasive species to restore natural biological diversity on Alaska Maritime National Wildlife Refuge

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Abstract The Alaska Maritime National Wildlife Refuge encompasses over 1.9 million hectares and more than 2500 coastal islands in Alaska. Like many other islands in the world, many refuge islands have had accidental and intentional introductions of non-endemic mammals (e.g., Arctic and red fox, ground squirrel, Norway rat, house mouse, caribou, reindeer, cattle, Arctic and European hare) that have drastically altered these fragile insular ecosystems. Although new introductions are prohibited, accidental introductions, particularly of rodents, are still of great concern. As part of a programme to restore native biological diversity, refuge personnel have surveyed most islands for exotic species, evaluated impacts of invasive wildlife on native birds, employed predator eradication methods, and assessed benefits of successful eradication. This paper reviews the history of these projects; particularly the effort resulting in eradication of introduced foxes from 39 islands totalling more than 500,000 hectares.

Keywords Aleutian Islands; Arctic fox; *Alopex lagopus*; island ecosystems; predator control; caribou; *Rangifer tarandus*; Norway rat; *Rattus norvegicus*; red fox; *Vulpes vulpes*; restoration; seabirds; ground squirrel; *Spermophilus parryii.*

INTRODUCTION

Impacts of alien species introductions to island ecosystems all over the world are well documented (e.g., Moors and Atkinson 1984). In spite of isolation and a harsh environment, Alaskan islands have not escaped accidental and intentional introductions (Elkins and Nelson 1954; Manville and Young 1965; Jones and Byrd 1979). Until about three hundred years ago, no terrestrial mammals were found on most islands (e.g., the central and western Aleutian Islands) south of the winter ice pack and isolated from the mainland since the Pleistocene (USDA Bureau of Biological Survey 1938; Murie 1959; Hopkins 1967; Tikhmenev 1978; Bailey 1993; Liapunova and Fedorova 1994). Surrounded by highly-productive seas, many of these islands provided nesting sites for large populations of marine birds. The native peoples of the region generally did not move terrestrial mammals around, and the islands retained largely intact faunas until Vitus Bering's discovery voyage in 1741.

The first deliberate introductions occurred soon after 1741, when Arctic foxes (*Alopex lagopus*) and red foxes (*Vulpes vulpes*) were released on several islands in the Aleutian chain (Black 1984; Jason 1985; Bailey 1993). The heyday of fox ranching occurred between 1910 and 1940 when nearly every habitable Aleutian island (about 86) was stocked with foxes, except for a few islands either too small or too rugged for regular access by wooden boat. To supplement the food available for foxes, particularly after bird populations declined, some trappers stocked islands with rodents such as ground squirrels (*Spermophilus parryii*) to the further detriment of native bird species (Peterson 1967; Janson 1985; Bailey 1993). New island residents brought with them livestock such as cattle, sheep, goats, horses, reindeer and bison. By the 1940s, most islands had some species of mammal introduced (Bailey 1993). After the Second World War, caribou (*Rangifer tarandus*) were purposely introduced to Adak Island (then a naval base) for sporting purposes (Jones 1966).

The earliest recorded accidental mammal introduction was prior to 1780 when Norway rats (*Rattus norvegicus*) became established on Rat Island from a Japanese shipwreck (Brooks 1878; Brechbill 1977; Black 1984). Norway rats became established on at least 16 other islands within the refuge over the last 200 years (Bailey 1993).

Despite reduction of native bird populations on islands with foxes and rats, enough islands remained free of terrestrial mammals in the early 20th century to draw attention of conservation-minded people to their wildlife values. Between 1909 and 1913, nine different islands or island groups in Alaska were set aside as National Wildlife Refuges, including the Aleutian Islands where many introductions of exotics had occurred.

Biologist O. J. Murie (1936, 1937) visited the region in 1936 and 1937; and he reported to government policy makers in Washington, D.C. on the decline of seabirds on refuge islands with foxes. This changed government policy on the use of the islands for fox ranching. Soon after WWII, Robert Jones (the first resident manager of the Aleutian Islands NWR) began eradication of introduced foxes.

The Aleutian Islands, other island refuges, and additional islands not previously designated as refuges were consolidated in 1980 to become the Alaska Maritime National Wildlife Refuge (AMNWR) under the Alaska National Interest Lands Conservation Act authority. The AMNWR refuge boundaries encompass 1.9 million hectares and over 2500 islands around the coast of Alaska. Few islands are greater than 2000 km^2. The primary purpose of the new refuge was to conserve (and restore where necessary)

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populations of marine birds, marine mammals, and terrestrial endemics. Most of the refuge land is undeveloped, uninhabited and stable except for some scattered communities, military bases, and abandoned cattle ranches and fox farms. Loss of habitat due to human development is not as great a threat in this region as is habitat degradation and conversion caused by invasive species.

The refuge islands discussed in this paper extend from west of Kodiak Island along the Alaska Peninsula and throughout the Aleutian Islands between the Bering Sea and the Gulf of Alaska. These islands range from slightly above sea level to steep glaciated volcanoes over 1900 m. Soils form in volcanic ash or cinders over basaltic rock, and higher elevations sometimes are covered in bare rock and basaltic rubble. Vegetation types change with elevation from coastal lowland bands of grass-covered dunes backed by herbaceous meadows to dwarf shrub communities (e.g. *Salix* spp. and *Empetrum nigrum*) in higher exposed areas. Precipitation (from 530 mm to 2080 mm) varies between large and small islands, and coastal and inland areas.

Restoration of native biological diversity by removing introduced predators and preventing additional accidental introductions is a major priority of the refuge. Foxes, ground squirrels, cattle, reindeer, caribou and rats are invasive species of primary refuge management concern. These species directly interfere with native birds through predation or loss of nesting habitat because of vegetation changes caused by overgrazing and trampling. Other exotic animals currently inhabiting refuge islands, but having minor or unknown impacts, include house mice (*Mus musculus*), deer mice (*Peromyscus* sp.), voles (*Microtus* spp.), hares (*Lepus* spp.), and hoary marmots (*Marmota calagita*). This paper summarises issues involving the major invasive species and how the refuge responds to the challenges that they pose.

FOXES

Distribution

Foxes were introduced to islands by Russians and by Aleut fox ranchers. Foxes were released on more than 450 Alaskan islands (Bailey 1993). Red foxes occur naturally on some near-shore islands along the Alaska Peninsula, and on the Fox Islands in the eastern Aleutians. Rainforests on Southeast Alaska islands provided poor habitat for red foxes, and they typically did not survive to become self-sustaining populations. Native to Bering Sea islands normally surrounded by sea ice (e.g. the Pribilofs, St. Matthew and St. Lawrence islands), Arctic foxes are apparently better adapted than red foxes to the Aleutian Island environment (Fay and Cade 1959; Chapman and Feldhammer 1982). Both species survived best on islands with tidal benches or accessible beaches for foraging. These habitats provide food during late fall and winter after migratory birds leave the islands (Stephenson 1970). Blue foxes (one morph of the Arctic fox) were more valuable than red foxes and eventually were placed on more islands than red foxes were. Introduced Arctic foxes currently remain on eight islands, which contain refuge lands, and introduced red foxes are still present on one refuge island.

Impact

Foxes severely reduce populations of nesting birds by eating eggs, nestlings and adult birds in summer and caching birds and eggs for later consumption. Particularly affected are waterfowl, shorebirds, seabirds, and ptarmigan (Bailey 1993). Vegetation also could be affected by the loss of fertilisation from large bird colonies.

Most of Alaska's breeding seabirds are not adapted to coexisting with terrestrial mammals. Almost all islands where introduced foxes persisted are treeless, so resident birds are particularly vulnerable since most species nest on the surface of the ground or in earthen burrows. For instance, foxes eliminated populations of Aleutian Canada geese (*Branta canadensis leucopareia*) on all but three islands, driving this endemic taxa close to extinction (Jones 1963; Byrd 1998).

Local residents (Aleuts and non-native fox ranchers) quickly recognised the impact of foxes on native birds, particularly the abundant seabirds. Most islands were stocked with only a few pairs of foxes (USFWS 1929-1939; Janson 1985), yet these introduced predators and their offspring quickly reduced populations of birds (Murie 1936; 1937; Swanson 1982; Black 1984).

Restoration

The staff of AMNWR began eradicating foxes from islands in 1949. To date (2002) foxes have been removed from 39 islands totalling more than 500,000 ha. There are plans to eradicate foxes from at least four more islands. Eradication efforts were hampered by federal regulations prohibiting toxicant use after 1972. Nevertheless, progressively larger islands have been cleared using only traps, firearms, snares, and M44 devices; in 1999 we exterminated foxes by trapping from a 90,000 ha island. We plan to eradicate foxes on smaller remaining islands without toxicants. Ebbert (2000) compares methods used to eradicate foxes on small and large islands.

Response

The response of native bird populations to fox removal has rarely been quantitatively documented. Nevertheless, it seems that populations of waterfowl, shorebirds, ptarmigan, seabirds, and possibly passerines increase following fox eradication, and without release of captive-reared birds or translocations (Williamson and Emison 1969; Day *et al.* 1979; Nysewander *et al.* 1982; Zeillemaker and Trapp 1986; Byrd *et al.* 1984; Byrd *et al.* 1997). Most populations of nesting seabirds increased at least four to five folds within 10 years of fox removal (Byrd *et al.* 1994). Initially, restoration of the endangered Aleutian Canada goose required transplanting geese to fox-free islands (Byrd 1998), but their populations have now increased from less than 1000 birds in 1975 to more than 35,000 in 2000 due to fox eradication (Byrd 1998).

RATS

Distribution

The adaptable Norway rat is established as far north as Nome, Alaska (65 degrees north latitude). Norway rats have become established on more than 16 islands within the refuge (Bailey 1993). About five of these sites were occupied by the military during WWII where numerous cargo ships unloaded supplies from ports where rats were prevalent. Ship rats (*Rattus rattus*) also became established for a time on one island occupied by the military, but were confined primarily to buildings and apparently disappeared when most buildings were removed (Taylor and Brooks 1995).

Impact

Rats extirpate most species of burrow-nesting seabirds (e.g., storm petrels *Oceanodroma* spp., Cassin's auklet *Ptychoramphus aleuticus*, tufted puffin *Fratercula cirrhata*), and they probably reduce populations of shorebirds (e.g., rock sandpiper *Calidris ptilocnemis*, black oystercatcher *Haematopus bachmani*) and other ground-nesting species. The probable result of rats becoming established on refuge islands used by colonial waterbirds is the eventual destruction of fossorial, crevice-nesting, and accessible surface-nesting seabird colonies as well as drastic reductions of certain species of other ground-nesting birds. On islands with introduced foxes, rats probably provide supplemental winter food, which keeps fox populations relatively high and thereby increases the impact of foxes on native birds during the breeding season. Some refuge islands (e.g., the Pribilof Islands) have endemic small mammals that may be vulnerable to predation and competition by Norway rats.

Prevention

Chances for accidental rat invasions on additional refuge islands are increasing now that fisheries, coastal tourism, and human population are increasing in Alaska. Once rats become established on islands larger than a few thousand hectares, removal is difficult and expensive. Rapid response following shipwrecks is needed to kill rats while they remain on the ship or as they come ashore. The refuge is prepared for such an eventuality (USDI 1993). Planned action involves local dispersal of single-dose baits adjacent to a grounded wrecked vessel or on the vessel itself.

Expanding on-shore fish processing development in the Bering Sea has recently resulted in new fish plants on two rat-free islands in the Pribilof Islands. Both communities, assisted by AMNWR staff, established bait and trap stations at their docks to eliminate rats that may disperse from infested vessels. The refuge continues to assist the two communities in developing and implementing rodent preventative measures (DeGange *et al.* 1995). Furthermore, an education outreach is underway to inform shipping companies of the dangers of rats on their vessels.

GROUND SQUIRRELS

Distribution

At least two subspecies of Arctic ground squirrels occur on approximately 17 refuge islands. *Spermophilus parryii ablusus* was introduced in the Aleutian Islands, and *S. p. nebulicola* is found on islands south of the Alaska Peninsula (Dufresne 1946). George Steller, naturalist on the first Russian ship to sail in Aleutian waters, noted in the Shumagin Island group that ground squirrels were present on islands near the Alaska mainland, but not on those farther offshore (Golder 1925; Stejneger 1936).

Ground squirrels were used by Native Alaskans and early Russians for clothing (parkas) and food and were transplanted on some islands from the mainland or islands where they were naturally occurring (L. Black, pers comm.). Ranchers also introduced ground squirrels to some islands as food for foxes, after seabirds declined (Peterson 1967; Swanson 1982; Janson 1985). Many introductions were not documented, and it is unclear whether some current populations on islands close to the mainland are native or introduced (Bailey 1993). Preliminary genetic analysis on squirrels collected in the Shumagins shows minor differentiation among islands (J. Cook, pers. comm.), but more samples are needed to determine if any island populations are unique.

Impact

Certain species of ground squirrels prey on eggs (Errington and Hamerstrom 1937; Horn 1938; Stanton 1944; Sowls 1948; Bedard 1969; Leschner and Burrell 1977; Sargeant and Arnold 1984; Sargeant *et al.* 1985). Arctic ground squirrels in research enclosures pounce on large duck eggs but are apparently not adept at opening them (B. Barnes, pers. comm.). Nevertheless, this species is known to take passerine eggs (B. Barnes, pers. comm.) and chicks and eggs of seabirds (Geist 1933; Cade 1951; Sealy 1966). The impact of introduced Arctic ground squirrels on nesting birds in AMNWR is not well documented, but we have observed that storm petrels and other small burrow-nesting species rarely occur on islands inhabited by ground squirrels.

Ground squirrels also directly affect native vegetation by feeding on stalks, stems and seeds, and contribute to overgrazing and erosion (Bailey and Faust 1981; Bailey and McCargo 1984). There is still a need to collect and analyse information to understand the biology and impact of non-native Arctic ground squirrels on refuge islands.

Restoration

We are at the beginning stages of devising or modifying existing control methods to eradicate introduced ground squirrels on 1450 ha Kavalga Island in the central Aleutians. Elsewhere in the United States, other ground squirrel species are controlled for agricultural purposes using poison baits, fumigants, trapping and shooting. Poison baits seem most practical for use on islands within the refuge. New registration may be required to use existing or new toxicants on AMNWR.

Potential non-target species that are the most likely scavengers of dead or dying ground squirrels include bald eagles (*Haliaeetus leucocephalus*), gulls (*Larus* spp.), and common ravens (*Corvus corax*). We may need to collect and dispose of ground squirrels dying above ground to minimise secondary poisoning hazard.

Predicted Response

Vegetation reduced by ground squirrels will likely recover rapidly and erosion of overgrazed areas will be slowed. Productivity will increase the first season after ground squirrels are eradicated for bird species on which egg or chick predation has been severe. We predict species that nest in earthen burrows such as storm petrels and surface-nesters such as shorebirds and passerines will benefit most from introduced ground squirrel eradication.

LARGE UNGULATES

Distribution of reindeer and caribou

Reindeer (*Rangifer tarandus asiaticus)*, a native ungulate of Eurasia, were brought to Alaska from Siberia in 1891. Reindeer were introduced on several islands in the eastern Aleutians to provide Alaska natives a commercial commodity to enhance their economic inclusion in the territory's development (Swanson and Barker 1992). Ultimately, reindeer were introduced to six islands that are now part of AMNWR.

Caribou (*R. t. arcticus*) are native to mainland Alaska but occasionally swim to nearby large islands. The only introduction in the refuge is on Adak Island in the central Aleutians where caribou were purposefully released in the late 1950s at the request of the U.S. Navy (Jones 1966). They have persisted, and the herd was recently estimated at approximately 900 animals (Williams 1998).

Distribution of livestock (cattle, horses, sheep)

Although a few cattle were brought to some of the islands during the fox ranching era, cattle ranching did not begin on most Alaskan islands until after WWII. Cattle ranching has occurred on at least eight islands containing refuge lands, but currently cattle occur on one refuge island and three other islands with both refuge and private lands. These same islands also have horses, and one has sheep.

Impact of introduced ungulates

Grasses and other flowering plants provide summer and fall foods for ungulates, but it is during winter that food becomes limited. Reindeer detect lichens through as much as 1m of loose snow and reach them by pawing. Reindeer "crater" into mineral soil while foraging on roots of forbs, dislodging plants, causing more severe damage. Introduced ungulates on relatively-small islands typically overgraze, damage the vegetation communities, and sometimes starve to death (e.g., Klein 1968). The first reindeer released on two islands in the eastern Aleutians died out from starvation relatively soon after stocking (Brickey and Brickey 1975), but later stockings have resulted in herds that have persisted. For reindeer on Alaskan islands, the common pattern has been rapid population growth resulting in depletion of native forbs, especially lichens (Swanson and Barker 1992). During overgrazing, more willow stem is consumed than can grow each season, and the plant eventually dies. When less-preferred forage is depleted before or during winter, the population crashes because of starvation (Palmer 1945). If the herd does not die out completely, the island does not sustain previous population levels due to long-term damage to lichens. Lichens may take 20 years or more to recover (Palmer 1945). Severe overgrazing and trampling by reindeer on preferred hilly areas also cause soil erosion and permanent loss of natural plant communities, reducing natural biological diversity and, in some cases, causing desert conditions. Typically, livestock are allowed to overgraze, and frequently cattle are abandoned because of difficulties in bringing them to market. Selective grazing by cattle makes sandy coastal areas especially vulnerable to damage (Talbot *et al.* 1984), and cattle ranching has led to the establishment of invasive plant species (Daniels *et al.* 1998).

Restoration and response

In most cases, reindeer, caribou, and livestock occur on islands with mixed ownership (usually with native villages) within the refuge. Native traditional councils typically manage reindeer for commercial meat and antler production and sometimes for sport hunting. Cattle grazing has been allowed by special permit. Ideally, introduced ungulates would be excluded from refuge properties, but there is no current plan or funds to erect and maintain fencing. Presently, the refuge staff works with local traditional councils, other government agencies and with permittees to develop management plans to minimise the negative impacts of grazing, trampling, and erosion. Swanson and Barker (1992) reviewed the history and range conditions of reindeer populations on Alaskan islands.

In the past, the refuge staff has removed reindeer and cattle from wholly-owned refuge islands, but these actions caused controversy and resulted in hard feelings by some local residents. For example, in the early 1990s, reindeer were removed from Hagemeister Island by live capture and shooting after the permittee failed to comply with maximum stocking rates, and damage was evident (Swanson and LaPlant 1987). At great public expense, 450 reindeer were live captured and transported to a nearby native village and the others were shot (most carcasses distributed to village for food). Nevertheless, the press provided substantial negative publicity (J. Stroebele, pers. comm.). In another case in 1985, substantial funds were spent to try to capture and transport feral cattle off Simeonof Island to nearby private locations before the remaining animals were shot. Again negative publicity occurred. In spite of poor public relations these management actions were necessary to restore native biological diversity. Response of native vegetation to removal of ungulates has occurred, and substantial restoration of some plant communities appears likely (S. Talbot pers. comm.).

Sport hunting kept the caribou populations at Adak under reasonable control until the number of residents on the island dropped from more than 5000 to less than 500 in the early 1990s because of the US Navy base closure. The caribou population is now expanding rapidly and habitat damage is inevitable (USDI & ADFG 1994). The island is now in joint ownership by the refuge and Native Village Corporation. Therefore, complete removal of the caribou is unlikely.

OTHER INVASIVES

Distribution

Bailey (1993) lists introductions of rodents and hares on refuge islands. House mice are known to live on St. Paul Island only in the community area and dump. Introduced voles are found on at least two islands, both south of the Alaska Peninsula, and marmots on another. Hares were released on at least 10 islands and still survive on a few. Deer mice occur on one island.

Impact

Although there is little documentation that the species listed above pose significant threats to native species on AMNWR, they probably negatively affect native forms. Small mammals negatively change native vegetation and habitat in other areas. Native voles are a significant predator of eggs and nestling parakeet auklets on St. Lawrence Island (Sealy 1982), and introduced voles may be responsible for sparse vegetation on small islands in the Sanak Group (E. Bailey, pers. comm.).

Deer mice are significant egg predators and severely limit reproduction of Xantus murrelets (*Synthliboramphus hupoleucus*) on Santa Barbara Island in California (Murray *et. al.* 1983) and elsewhere (Maxon and Oring 1978).

Restoration and response

Currently no plans exist to remove any small mammal species referred to in this section, although we may attempt to eradicate other non-native rodents on islands where rat or ground squirrel eradication is planned. An evaluation similar to that described for ground squirrels is needed prior to eradication efforts.

DISCUSSION

Implications for work

Because of the controversial nature of eradication projects in the United States, results of the fox eradication programme are not widely published. Now that the red fox eradication programme nears conclusion, publication of methods used and results achieved is more appropriate. Lessons learned during this project are applicable to other island restoration projects.

The U.S. Fish and Wildlife Service does not permit introduction of exotic species on refuge lands. The State of Alaska's wildlife laws are also stringent. Refuge islands remain some of the most inaccessible and least-visited islands in the U.S. National Wildlife Refuge System. The islands are fairly safe from deliberate introductions; however, a constant vigil is needed to prevent the accidental introduction of rats through shipwreck or transfer of infested material.

Suggestions for improvement

To document the benefits of removing invasive species, biological monitoring must continue on the few islands where pre-eradication data exists. Widespread genetic sampling of Arctic ground squirrels is necessary to determine the native status of island populations before eradication can proceed. To efficiently eradicate ground squirrels and other rodents from refuge islands, development of new baiting strategies or the modification of existing methods are needed. Modification of existing pesticide registration for island applications is also necessary.

Public education and acceptance of the need to remove large herbivores, such as cattle and reindeer, from islands must be accomplished before further eradication of these species can proceed. Even if methods are devised to efficiently eliminate these large, familiar and once-domesticated animals, the project could not be considered successful if it results in long-lasting damage to trust and public perception of the AMNWR and its mission.

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Control and eradication of the introduced grass, *Cenchrus echinatus*, at Laysan Island, Central Pacific Ocean

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Abstract The sandbur, *Cenchrus echinatus,* an annual grass native to Central America, was first documented occurring at Laysan Island, Hawaiian Islands National Wildlife Refuge, in 1961. The military or researchers visiting the island probably took it there inadvertently. By 1991 it had spread to become the dominant species in some 60 hectares or 30% of the vegetated area of the island. By displacing the native bunchgrass, *Eragrostis variabilis*, it diminished important breeding habitat for two endemic, endangered landbirds; the Laysan finch (*Telespiza cantans*) and the Laysan duck (*Anas laysanensis*), as well as several species of indigenous seabirds and terrestrial arthropods. In 1991 Refuge staff started a year-round control programme designed to eradicate *Cenchrus echinatus*. After experimenting with a range of techniques including heat and saltwater application, we found application of a herbicide (glyphosate) and mechanical control (hand pulling) to be most effective. Concurrent studies of the life history of the plant allowed continual adjustment and refinement of the eradication programme. Decline in the rate of finding new plants in a previously-cleared plot from as high as 85 plants per hour in Autumn 1994 to 0.043 plants per hour (or one plant per 23 hours searching) in Autumn 1999 is evidence that the seedbank is being depleted. *Cenchrus* is now so rare that it no longer has effect on the ecosystems of the island. Costs for the project include a monetary investment averaging US USD150,000 per year for staff, supplies, and vessel charter to this remote site (five days by boat from Honolulu); disturbance to nesting seabirds, and the risks of introducing new island pests despite stringent quarantine procedures.

Keywords invasive; restoration; glyphosate

INTRODUCTION

Laysan Island is a 411 hectare island in the north-western Hawaiian Islands at latitude 25°42'41"N and longitude 171°44'06"W. It was declared part of the Hawaiian Islands Bird Reservation by presidential order in 1909 and today makes up part of the Hawaiian Islands National Wildlife Refuge, managed by the U.S. Fish and Wildlife Service for the conservation of natural ecosystems and the protection and recovery of endangered species and migratory birds. The refuge consists of basalt islands, coral islands, atolls, and reefs; most of which are uninhabited. It stretches over 1370 kilometres to the north-west of the main Hawaiian Islands in the Central Pacific Ocean. Even though these remote islands were set aside for conservation relatively early, they did not escape all the exploitation and biological invasions to which oceanic islands are particularly vulnerable. Between 1887 and 1915 guano mining and feather hunting caused major disruption to the island's ecosystem (Spennemann 1998; Ely and Clapp 1973) but the most profound damage came after rabbits (*Oryctolagus cuniculus*) were deliberately introduced around 1903. The subsequent defoliation of the island due to this population of rabbits extirpated an unknown number of terrestrial invertebrates and plants, and caused the extinction of three land bird species before the last rabbit was killed in 1923. Scientists on the expedition in 1923 and again in 1930 replanted various plant species, some indigenous and some not (summarised in Ely and Clapp 1973 and Newman 1988). Today there are 17 native and 14 introduced plants on Laysan Island.

In 1961 biologists first detected *Cenchrus echinatus* or sandbur on Laysan Island. They killed *Cenchrus* plants found on that expedition but some survived, and by 1984 the species had spread to occur in 22 of 161 randomly located plots on Laysan Island or in 14.6% of sites (Newman 1988). At the peak of the infestation in 1991 *Cenchrus* grew on an area of 63.6 hectares representing 30% of the 212 vegetated hectares of the island. In 1990 the refuge manager decided the rapid spread of *Cenchrus* posed a threat to the health of the habitat and wildlife populations at Laysan and committed resources to a programme to eradicate the grass from the island. He chose to concentrate efforts on *Cenchrus echinatus* rather than the 13 other non-native plant species because this annual grass was obviously changing the ecosystem of Laysan Island. It seeds prolifically, forms mats, and it appeared to be displacing the native bunchgrass *Eragrostis variabilis* over large areas of the western part of the island. *Eragrostis* is a perennial bunchgrass and the dominant species on Laysan. It was seen in 117 of the 161 plots (77.5%) surveyed by Newman in 1984 (Newman 1988). This species is used by almost all the bird species breeding on Laysan as nesting habitat and cover. Of particular concern is its importance to the two endemic species of landbirds listed as endangered under the U.S. Endangered Species Act, the Laysan duck (*Anas laysanensis*) and the Laysan finch (*Telespiza cantans*). The Laysan duck prefers to nest deep within the clumps of *Eragrostis variabilis* (Moulton and Weller 1984). The Laysan finch at Laysan Island nests almost without exception in clumps of *Eragrostis* (Morin

Pages 110-115 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

and Conant 1990). The finches also eat the seeds of the bunchgrass. *Eragrostis* is also important for most of the 17 species of breeding seabirds at Laysan that nest on or under the ground by providing cover for nests and giving structural stability to the soil to prevent burrow collapse. Whereas *Eragrostis* continues to provide cover and retains its physical structure even after it dries up, the *Cenchrus* leaves almost nothing when the plant dies.

The objectives of the management programme initiated in June 1991 were to locate, map, and kill all *Cenchrus echinatus* on Laysan. A regime was then established in which all areas could be visited and cleared of newly sprouted *Cenchrus* before seeding could occur. The manager made a commitment to continue this until the seed bank was completely depleted and eradication achieved. In addition to plant control, the staff monitored the plant community to assess progress and effects of the vegetation management and measured aspects of the life history of *Cenchrus* in order to refine control methods. Prior to the eradication programme the staff usually visited Laysan once per year. In June of 1991 a field camp was established that has been continuously occupied by at least two biological technicians since then.

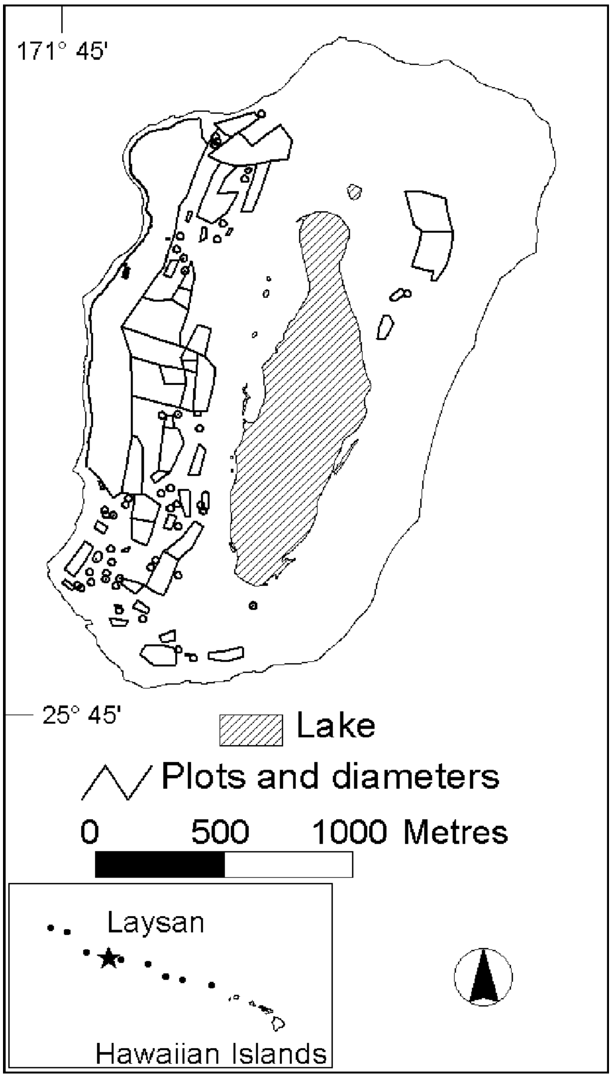

Fig. 1 Laysan Island, north-western Hawaiian Islands. Locations of plots and diameters in which *Cenchrus echinatus* was removed. The largest plot on the west side of the island was named the Blob.

METHODS

Control and Eradication

The staff tested several methods of killing *Cenchrus*, including heating plants with a propane torch, applying salt water, mechanical removal, and herbicide application. The most effective method for killing large mats and big plants with the least collateral damage to wildlife was to spray with 1% glyphosate (RODEO) mixed with a surfactant (LI700, Loveland Industries, Inc.) and a dye (Turf Mark, J. R. Simplot, Co.) to indicate areas already sprayed. We brought the large amounts of water needed to mix the herbicide by ship at each camp re-supply trip (three times per year) until 1994 when we installed a solar-powered reverse osmosis water maker. We applied herbicide using hand-pumped backpack sprayers. All sites at which any *Cenchrus* was found were assigned a permanent number, marked with posts made of 1.27 cm PVC pipe, and mapped. We marked patches of *Cenchrus* containing many plants along the entire perimeter and assigned a plot number. We marked the sites of solitary plants with a single pole and assigned a diameter number. These plots and diameters were placed in a rotation schedule in which they were revisited at intervals designed to allow detection of newly-sprouted *Cenchrus* before it could seed. The objective was to keep all plants in an area from setting seed and eventually deplete the seed bank and break the cycle of growth. Figure 1 shows all plots and diameters in which *Cenchrus* has been found and which are visited according to schedule. Greatest effort was first concentrated on plots located furthest to the north-east because the prevailing wind at Laysan is from that direction.

After initial spraying the technicians pulled subsequent regrowth by hand and removed it from the site in plastic bags. They brought the small amounts found to camp and burned them in a barrel. The main job in most plots after clearing of initial distribution of the weed becomes careful scrutiny of the entire area for any sign of *Cenchrus* sprouts. We initially set the interval between checks at two weeks but as life history data were collected we determined that *Cenchrus* did not go to seed for 8-19 weeks from initial sprouting. We changed the rotation schedule to increase visitation intervals to once every six weeks for three years after the last *Cenchrus* plant was found and then once per 16 weeks thereafter if no new plants were found in that time. Finally if a plot or diameter had no new plants found in five years, we changed the plot visitation rate to once per year. If a new plant appeared in a very large plot, a new interior diameter was established at the site to preclude having to increase the visitation rate for the entire plot. At every visit the staff recorded time spent in the plot, number of *Cenchrus* plants found and their stage of development, and number of seabird burrows crushed. In 2001 as time constraints were eased from the assignment of more and more plots to the least frequently visited category we created one more category. "Hotspots" are plots or diameters in which *Cenchrus* has been found within the last two years and in two consecutive visits. We visit these sites every four weeks.

We overlaid a 100 metre grid on a map of the entire island and systematically searched all sections as often as possible (usually once every two years) to locate new or undiscovered patches of *Cenchrus* as well as to maintain surveillance for other previously-undetected weeds.

The extremely high density of wildlife (hundreds of thousands of breeding seabirds) present year-round at Laysan Island necessitated special measures and considerations to reduce impacts. Spraying herbicide instead of pulling minimised the time the staff spent in any particular part of the colony and protected burrow structure. We chose glyphosate because of its relatively low vertebrate toxicity. We restricted the height of the PVC pole markers to reduce the collision hazard for flying seabirds. Early in the evaluation of methods for killing *Cenchrus echinatus* we rejected the idea of pursuing a biological control agent due to possible presence of an indigenous congener, *Cenchrus agriminioides laysanensis.* We believe that this variety is now extinct but did not choose to take the risk of introducing a biological control agent that might harm any survivors. When plots and diameters moved into the once-per-year rotation schedule we visited them during November and December when the fewest burrow-nesting seabirds were present and vulnerable to burrow collapse.

Maintaining a year-round camp increased the number of people visiting and the importation of equipment and food. This raised the probability of introducing new plants, insects, and pathogens to Laysan Island. A strict quarantine protocol has been in effect for the duration of the project: this includes the requirement that all soft gear (clothing, shoes, tents, packaging, etc.) that is brought to Laysan be brand new. All goods are packed in plastic containers. No corrugated cardboard is permitted. All items except for electronic and optical gear must be frozen for 48 hours prior to being landed at Laysan.

Monitoring

Throughout the operation we monitored the plant community and the resident bird community to assess the extent and effects of the *Cenchrus* infestation, to refine control techniques, to measure success in control efforts, and to measure the effects of control efforts on other species of plants and animals. We also measured standard weather variables (rainfall, temperature, wind velocity, cloud cover) to assess their relationship to *Cenchrus* growth.

Plant Community

In 1989 the staff established five vegetation transects to monitor the spread of the largest of the *Cenchrus* distributions (the Blob). These transects were lines ranging between 150 and 300 m long and placed to be perpendicular to the boundary between the interior of the Blob and the *Cenchrus*-free areas adjacent to them. Twice per year we recorded plant species or substrate at each metre mark on tape extended between the two ends.

We studied *Cenchrus* life history by marking individual sprouts as they emerged and continuing to monitor their development. We checked plants once per week to record the age when seeds appeared and when seeds were mature.

Effects on seabird populations

We studied the effects of *Cenchrus* and *Cenchrus* control actions on avian populations by establishing twelve 10m x 10m plots on the west side of the island. Eight plots were located adjacent to each other in the heart of the main *Cenchrus* distribution. Four were located at a site of similar aspect and distance from the ocean but outside the area infested by *Cenchrus.* In the infested area we cleared four of the plots of *Cenchrus* and maintained them *Cenchrus*-free using the techniques standard to the rest of the island. The other four plots were allowed to remain infested. We measured percentage cover of *Eragrostis,* percentage cover of *Cenchrus*, percentage cover of other plant species, numbers of clumps of *Eragrostis*, numbers of all active and empty nest sites for all bird species and the contents of those nests, and numbers of all birds in the plots not associated with nests. This study continued from June 1991 to April 1995.

RESULTS

There are currently 90 plots of varying sizes and 161 circles (called "diameters") with a 15 metre radius. Of these, 24 plots and 136 diameters have moved to the once-per-year rotation because no *Cenchrus* has been found for more than five years. Forty-nine plots and 19 diameters have been moved to visits once per tour (~16 weeks) because *Cenchrus* has not been found there in at least three years. The remaining 17 plots and six diameters remain on a six-week rotation schedule. In the period from October 2000 to March 2001 two full time technicians found only 13 *Cenchrus* plants, five of those that had seeds. From March 2001 to July 2001 they found no *Cenchrus* plants.

By tracking individual *Cenchrus* plants the staff measured time required from sprouting to production of mature seeds. During the winter months (October-March) the first mature seeds appeared between eight and 19 weeks after marking (mean 12.3 weeks, n = 37). During a summer trial we observed comparable results with plants taking between eight and 12 weeks to produce mature seeds (mean 9.6 weeks, n = 6) (Marks 1995).

Cenchrus echinatus had a deleterious effect on wildlife by displacing the dominant plant *Eragrostis variabilis*. Figure 2 illustrates *Eragrostis* density in a transect through an area without *Cenchrus* and changes in percentage cover

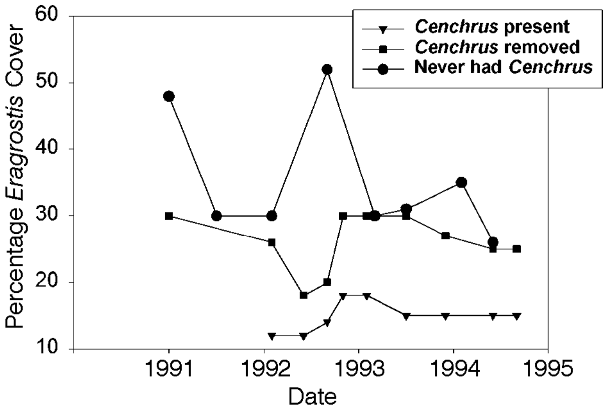

Fig. 2 Difference in *Eragrostis variabilis* cover between areas with and without *Cenchrus echinatus* present. Transect A was an area that never had *Cenchrus*. *Cenchrus* was removed and kept out in Plot B and left intact in Plot CB.

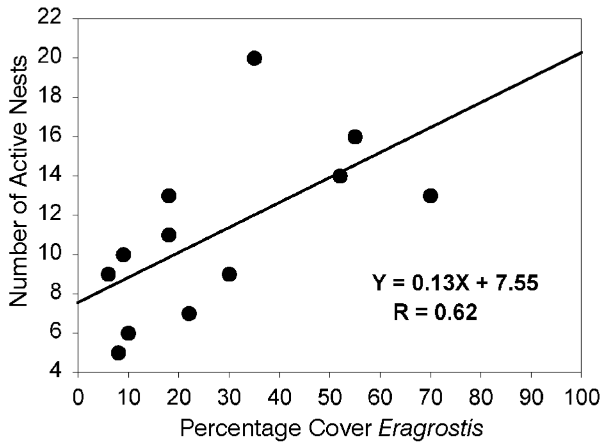

Fig. 3 Relationship between percentage cover of *Eragrostis variabilis* and number of active seabird nests in 12 plots (10 x 10 m) in July 1993. Y = 0.13X + 7.55, R = 0.62

of *Eragrostis* over a four-year period in two adjacent plots (B and CB). *Cenchrus* was removed and excluded in plot B and left intact in CB. Figure 3 illustrates the positive relationship between seabird nest density and percentage cover of *Eragrostis variabilis*.

Extirpation patterns and seed bank persistence remained consistent throughout the period in which plots were brought under the control regime. As an example the average number of *Cenchrus* plants found per hour and the average number of plants found in seven different plots over a two and a half year period are shown in Fig. 4. The numbers of plants found in a very large plot over a period of approximately three years from initial clearing (Fig. 5) may indicate that seed bank depletion is somewhat independent of environmental conditions such as temperature and rainfall. All areas of Laysan Island had extensive mixing and turnover of the soil through the digging actions of the burrow-nesting seabirds such as wedge-tailed shearwaters (*Puffinus pacificus*) and Bonin petrel (*Pterodroma hypoleuca*). This accelerated the rate at which seeds were exposed to conditions that triggered germination. A very efficient ally in the depletion of the seed bank was the Laysan finch (*Telespiza cantans*). These granivorous birds actively searched the soil for seeds and destroyed them as they consumed them. Decline in the rate of finding new plants in a previously-cleared plot from as high as 84.7 plants per hour in Autumn 1994 to 0.043 plants per hour in Autumn 1999 is evidence that the seed bank is being depleted.

The monetary cost of eradicating *Cenchrus echinatus* at Laysan was high due to the extreme remoteness of the site. Prior to the initiation of the project we managed the refuge by visiting only once per year so the necessity of establishing a year-round camp significantly increased the annual expenditures for this site to an average of USD150,000 per annum. Although we did other biological and management tasks while at the field site we can attribute the entire budget to the eradication effort because we would not have maintained a permanent camp there if not for the *Cenchrus* project.

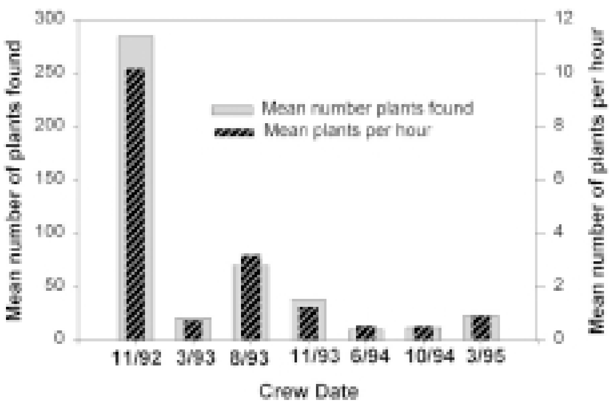

Fig. 4 Average Number of *Cenchrus echinatus* plants found per hour and average total plants found in seven different plots on Laysan Island between November 1992 and March 1995.

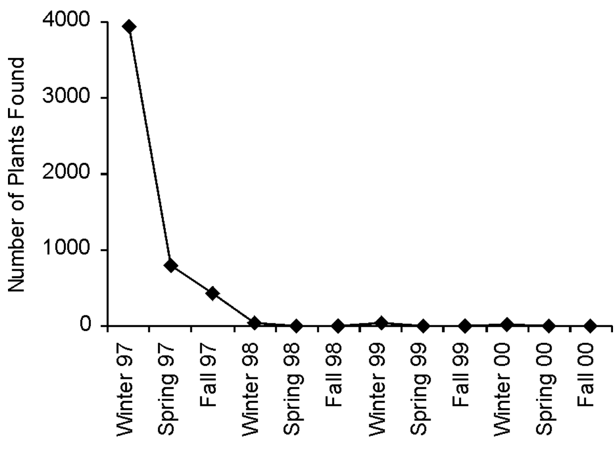

Fig. 5 Number of *Cenchrus echinatus* plants found in South Blob plot on each tour after initial clearing.

It is somewhat harder to quantify the impacts of an eradication program on wildlife. Hundreds of burrows of nesting seabirds were destroyed each year during operations. Some of these cave-ins killed the eggs or more rarely the chicks in the burrow. Small numbers of adult albatrosses and terns were also killed or injured in collisions with the radio antenna in camp or with the PVC plot poles. Grey-backed terns (*Sterna lunata*) and sooty terns (*Sterna fuscata*) flushed from their eggs by our activities lost eggs to ruddy turnstones (*Arenaria interpres*) and Laysan finches. Despite the adverse effects of our activities at Laysan on individual seabirds we do not believe that our work was detrimental to any population of birds at Laysan Island.

In the 10 years since we have had the quarantine protocols in effect, we have had only one possible introduction of a plant. A single seedling of what could only be tentatively identified as a member of the genus *Medicago* sprouted in 2000. It died before it flowered. Our ability to detect new terrestrial invertebrate introductions, and soil-borne pathogens is less well-developed due to less information about the invertebrate community at the outset of the project but we are not aware of any introductions that can be attributed to the *Cenchrus* eradication.

DISCUSSION

A year-round effort targeting the invasive grass *Cenchrus echinatus* at Laysan Island was successful at reducing the plant to almost undetectable levels. In balancing between active management and scientific documentation of the biology of *Cenchrus* and the effects of its removal, we put most resources into plant removal rather than into exhaustive monitoring of outcomes or analysis of data.

In 1993 refuge staff discovered a small, incipient invasion of *Cenchrus echinatus* at Rose Atoll, American Samoa. When found there were 10 robust clumps, most of which had gone to seed. These plants were pulled and the site visited again on subsequent trips. The seeds remaining in the soil had sprouted by the next visit in 1994. Biologists working with the Department of Land and Natural Resources of the American Samoa Government pulled all the plants again, burned the area, and covered it with a plastic tarpaulin. This action killed the remaining seeds in the seed bank and today Rose Atoll is *Cenchrus*-free. The Rose Atoll experience underscores the value of early intervention, especially at sites that you cannot occupy year-round.

The cost per plant of continuing the *Cenchrus* work each year has become very high but the cost of discontinuing the programme too soon is also very high. The probability that the eradication of *Cenchrus* will be successful at Laysan Island is higher than might be predicted for other infestations of comparable size because of the extreme isolation of the site, the high degree of control the management agency has over access to the island, and the ability to maintain the effort throughout the long period of extremely low *Cenchrus* yield that inevitably occurs at the end of any eradication effort.

With the imminent extirpation of *Cenchrus* at Laysan, the staff have incorporated other restoration activities into their schedule including propagation and planting of indigenous species that either had become exceedingly rare at Laysan (*Mariscus pennatiformes*, *Chenopodium oahuense*) or were completely eliminated by rabbits prior to 1923 (*Pritchardia remota, Capparis sandwichiana*, *Santalum ellipticum*). The decision to control or attempt eradication of other non-indigenous species at Laysan will be made on an individual basis when our understanding of each species' role indicates that it has the potential to have the same profound effects observed during the *Cenchrus echinatus* invasion.

ACKNOWLEDGMENTS

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The eradication of *Rattus rattus* from Monito Island, West Indies

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Abstract Monito Island (15 ha) is located between Puerto Rico and Hispaniola (West Indies). The island is inhabited by the endemic Monito Island Gecko (*Sphaerodactylus micropithecus*), which is scarce and exhibits a restricted distribution. Rat (*Rattus rattus*) predation has been postulated as the most likely explanation for this. The Puerto Rico Department of Natural and Environmental Resources (PRDNER) started a rat eradication programme on Monito in October 1992, using Maki Mini Blocks®. Rodenticide was spread at 10 m intervals over the entire island. Rats declined from a relative estimate of 0.63 to 0.01 rth (rat/trap hour). In April 1993, this project was stopped by the U.S. Fish and Wildlife Service due to concern about the possibility of poisoning geckos with the rodenticide. We proved experimentally that the geckos were not attracted to the paraffinised rodenticide blocks, but in the meantime there was a large increase in rat abundance to 0.28 rth. We started the project again in September 1998, when a surprisingly low rat population was barely detectable using chewing sticks. No rats were caught in snap traps, but the eradication programme went ahead, using Talon-G®. Rodenticide blocks were put in plastic baiting stations, spaced at 20 m intervals. Recent surveys have detected no rats on Monito Island. The rat eradication will have great benefits for Monito's unique biota.

Keywords Rodenticide; *Sphaerodactylus*.

INTRODUCTION

Monito Island (15 ha) is located at Latitude 18° 10' N and Longitude 67° 57' W (Wadsworth 1973), in the Caribbean Sea between Puerto Rico and Hispaniola (West Indies) (Fig. 1). It belongs politically to the Commonwealth of Puerto Rico, and is a unique component of a national system of natural reserves. Subtropical dry forest is the only life zone present in Monito (Ewel and Whitmore 1973). Rainfall is more abundant during September-November, and less abundant during February-April. This island is basically a flat plateau surrounded by vertical cliffs (66 m) with no beach. It is considered to be the most inaccessible island within the Puerto Rican Archipelago.

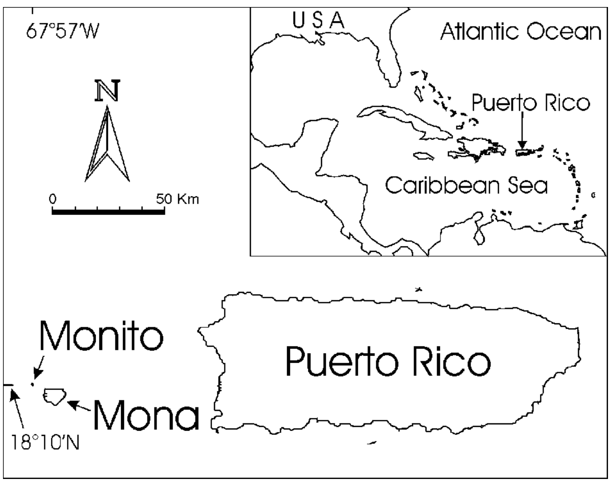

Fig. 1 Monito Island and its location in relation to Puerto Rico and other Caribbean Islands.

Monito Island harbours a unique fauna, including one of the largest seabird nesting colonies in the West Indies, and an endangered species of gecko (USFW 1986). Since its discovery in 1974, this endemic gecko, *Sphaerodactylus micropithecus* (Schwartz 1977) has been considered scarce and restricted in range to Monito Island. In fact, predation by the ubiquitous black rat (*Rattus rattus*) was postulated as the most possible explanation for the current status of this reptile (Dodd and Ortiz 1983). Rats have never been observed preying upon geckos in Monito. However, this rodent has caused the extinction or extirpation of several species of reptiles (Crook 1973; Whitaker 1973, 1978; Lever 1994), birds (See Atkinson 1985 for review; van der Elst and Prys-Jones 1987; Lever 1994), and invertebrates (Ramsay 1978; Howarth and Ramsay 1989).

In October 1992, the Puerto Rico Department of Natural and Environmental Resources (PRDNER) began an eradication programme for black rats on Monito Island, encouraged by the successful rat eradication on Cayo Ratones, La Cordillera Natural Reserve (PR), and on Steven Cay (US Virgin Islands). In both projects, rats were eradicated using anticoagulant rodenticides without affecting non-target species.

The first stages of the eradication campaign with rodenticide produced promising results. Nevertheless, in April 1993, this initiative was restricted to the use of snap traps by the United States Fish and Wildlife Service (USFWS). The USFWS claimed that the PRDNER had not satisfied all the requirements of the Federal Insecticide, Fungicide, and Rodenticide Act (FRIFRA). The major concern was the possibility of poisoning Monito Island geckos with the

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anticoagulant rodenticide. A previous preliminary experiment using a similar gecko species *Sphaerodactylus macrolepis*, resulted in 15% mortality after exposure to pellets of the anticoagulant rodenticide Talon-G® (Gaa 1986).

We conducted a study to test the effect of anticoagulant rodenticides on captive geckos. We used the Mona Island Gecko, *Sphaerodactylus monensis* as a surrogate species because it is very abundant, lives in a comparable habitat, and is similar in size and in feeding habits to the Monito Island gecko. The experiment was conducted over 22 days. Maki® (Liphatech, Milwaukee, USA) was utilised instead of Talon-G, because it can be purchased over the counter without a license to apply pesticides. Four treated and four control cages were used. Three geckos and two Maki Mini Blocks (bromadiolone 0.05%) were placed within each cage. The results were not statistically analysed since all geckos survived the experiment. We neither observed changes in the behaviour of the geckos which might be related to poisoning (e.g. erratic movements or immobility), nor saw geckos licking or eating the pellets of poison. We re-initiated the second eradication campaign in September 1998, this time, with the approval and commitment of both the USFWS and the PRDNER.

METHODS

1992-93 campaign

The first eradication campaign started in October 1992, when we spread 13 buckets (9.09 kg each) of Maki mini blocks. We distributed the Maki blocks throughout the island, following a grid design. The distance between each pair of grid points was 10 metres, and three to five blocks were deposited at each grid (i.e. baiting) intercept. The rodenticide was also freely dispensed in areas of high rat activity, such as bird rookeries.

We continued the eradication campaign in March 1993. Although we repeated the original methodology, this time we used 20 buckets of baits. We decided to increase the amount of rodenticide for the second event because Monito was topographically more complex than we had previously expected.

We used snap traps to assess and monitor changes in the rat population during the eradication campaign. Twelve snap traps were equally spaced on a 120 m trap-line. We trapped rats over three consecutive nights. Each trap was set around 1900 hours and then checked every hour until 2200. For bait, we used a combination of processed cheese food (i.e. Cheez-Wiz®) and oat flakes to add consistency. An estimate of rat relative abundance was calculated by dividing the total number of rats caught by the total number of hours that traps were set to catch rats (rat/trap hour). Sprung traps without rats were not included in this ratio. We evaluated the status of the rat population seven times during this first campaign. These surveys were conducted before, between and after the poisoning events.

1998-99 campaign

The second eradication campaign consisted of three poisoning events at four-month intervals. On each trip of the eradication, 30 buckets (5.45 kg each) of Talon-G (ICI Americas Inc. North Carolina, USA) were distributed over Monito Island. The first event was conducted in October 1998, following a survey of the island in September to evaluate the status of the rat population. During the first night of the survey no rats were caught in snap traps. We therefore shifted to chewing sticks as a monitoring tool to detect rats at apparently-low population density. Fifty chewing sticks were freely distributed throughout Monito. We marked each chewing stick location with a blue flag.

Blocks (6-8) of Talon-G (brodifacoum 0.05%) were then placed in baiting stations distributed at 20 m intervals forming a grid over the entire island. Baiting stations were used to extend rodenticide availability, increase the chances of consumption, and decrease the chances of poisoning non-target species. We built the stations using plastic (PVC) sanitary pipes (10.16 cm width x 24 cm length).

Once again we evaluated the effectiveness of the eradication every two months using snap traps. Ten snap traps were set every 10 m of each 100 m trap-line. We ran three trap-lines, following the same protocol used during the surveys of the first campaign. We determined the status of the rat population five times during this campaign. One survey was conducted prior to poisoning, two during the poisoning events, and two after.

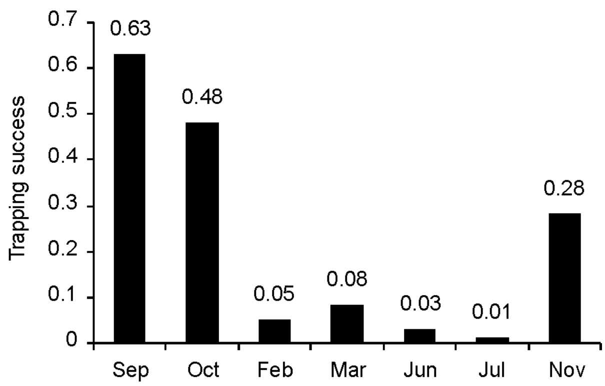

Fig. 2 Effectiveness of rat eradication measured as the number of rats caught per trap hour of effort (Sep. 1992-Nov. 1993). Rodenticide was spread in Oct. 1992, and Mar. 1993.

RESULTS

1992-93 campaign

The first index of rat population was 0.63 rat/trap hours (r/th) (Fig. 2). This catch rate decreased dramatically from 0.48 to 0.05 r/th after the first poisoning in October 1992 (Fig.2). Although the catch rates had risen slightly (0.08 r/th) by the time of the second poisoning in March 1993, it continued to decrease in the surveys of June 1993 (0.03 r/th), and July 1993 (0.01 r/th) (Fig. 2). Unfortunately, by November 1993 the rat catch rate had increased substantially (0.28 r/th) (Fig. 2). The rat population had multiplied to almost pre-poisoning values in only nine months without spreading rodenticide.

1998-99 campaign

During the rat survey in September 1998, only three of the 50 chewing sticks showed rat evidence. These three sticks were found in the south west area of Monito. We did not catch rats in snap traps during any of the five surveys conducted (640 trap hours). These surveys were performed in September 1998, October 1998, April 1999, August 1999, and September 1999. We spread rodenticide in October 1998, April 1999, and August 1999. Since the completion of the poisoning we have not detected the presence of rats in Monito Island. However, we have not yet used chewing sticks to assess the status of the rat population.

DISCUSSION

We found that the main difficulties in eradicating rats from Monito Island were related to the island's inaccessibility. However, with the help of a helicopter to transport the rodenticide buckets and the baiting stations this problem was greatly reduced. Monito Island lacks native terrestrial mammals or resident avian predators. Thus, the probability of secondary poisoning was minimal. The direct risk of poisoning non-target species, such as the endemic yellow-shouldered blackbird (*Agelaius xanthomus*) and the zenaida dove (*Zenaida aurita*), was reduced by the use of baiting stations.

The decline in rat abundance was unexpected between the first and the second campaign in Monito Island. This was especially surprising since during the first campaign, the rat population had increased dramatically in just nine months. After more than five years without poisoning (March 1993-September 1998) we expected to find rat abundance similar to that of 1992. A possible explanation is a prolonged drought. Although rainfall data from Monito does not exist, there are data from Mona Island, which is 5 km south-east of Monito. The monthly climate data does not reflect any lasting decrease in rainfall between 1993 and 1998 and the mean annual rainfall values do not indicate any abnormal reduction in rainfall: 5.72 cm (1998), 5.56 cm (1997), 8.46 cm (1996), and 5.87 cm (1995). Other potential explanations are a disease outbreak and predation. No data are available to support or negate a disease outbreak. With the exception of a pair of migratory peregrine falcons, there are no rat predators on the island.

Eliminating the rat's detrimental effect on Monito Island will undoubtedly have beneficial results for Monito's native and unique biota. To be certain that eradication has been achieved it is essential that the appropriate rat monitoring continues on Monito, especially using chewing sticks. Fortunately, if the eradication has been successful, the probability of re-infestation is extremely low due to the extreme isolation and rugged topography of Monito Island.

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Changes in bird numbers on Tiritiri Matangi Island, New Zealand, over the period of rat eradication

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Abstract Tiritiri Matangi is 25 km north of Auckland City in the Hauraki Gulf, New Zealand. Most of its forest cover was removed during many centuries of Maori and European occupation and farming. Some areas of extant forest canopy remained. Farming ceased in 1971. Since 1984 some 300,000 native trees have been planted. Twenty-seven species of native bird are naturally present and breeding on the island. Twenty-two exotic species introduced to mainland New Zealand have found their way to the island. Nine species of native bird have been translocated to the island. Data from bird counting transects within extant forest areas in spring are considered. The data from a three-year period before eradication of Pacific rats (*Rattus exulans*) in September 1993 are compared to a three-year period following rat eradication, with a three-year settling period between. A number of significant changes are recorded with both increases and decreases in bird numbers. These are attributed to the direct impact of the rats or changes in forest composition following rat removal, or the data are confused by conservation management action beyond the immediate count areas.

INTRODUCTION

Tiritiri Matangi is a low-lying 220 ha island lying 4 km off the Whangaparaoa Peninsula and 25 km north of Auckland City in the Hauraki Gulf, New Zealand. It is a Scientific Reserve under the Reserves Act 1977, and is open to public visitation.

Maori occupied the island prior to the arrival of Europeans in New Zealand, and from at least 1841 it was grazed by domestic animals. A lighthouse was established on the south-eastern end in 1865. The Crown withdrew the grazing lease in 1971, and management of the island was then taken up by the Hauraki Gulf Maritime Park Board. At that time it was proposed that, apart from the Lighthouse Reserve area, native vegetation be allowed to regenerate naturally.

In 1979 a programme of planting to enhance regeneration was proposed, with a plan which called for the planting of most of the island while leaving selected areas to regenerate naturally (Department of Lands and Survey 1982) (Fig. 1). In the period 1984 to 1993 more than 280,000 native trees were planted, increasing the proportion of non-grassland vegetation from 6% to 60% of the island's area (Galbraith and Hayson 1995). Some 20,000 trees have been planted since 1993, but planting has now ceased.

The Pacific rat or kiore (*Rattus exulans*) is presumed to have been on the island at the time of first European contact but was removed in an operation during September 1993 (Veitch 2002). Cats (*Felis catus*), rabbits (*Oryctolagus cuniculus*) and goats (*Capra hircus*) were reported as having feral populations that were subsequently removed (Dept. of Lands and Survey 1982, Moller and Craig 1987). Cats were probably never established as a feral population and the occurrence referred to by the Dept of Lands and Survey (1982), and later quoted by Moller and Craig (1987), related to domestic cats owned by a lighthouse keeper (A. Wright pers. comm.). Rabbits which were at one time plentiful had disappeared by 1908 (Dept of Lands and Survey 1982). The goat population was small and was removed by the lighthouse keepers. This work was under way in 1961 (A. Wright pers. comm.) and no goats were present in 1971 (R. Walter pers. comm.).

Seventy-seven bird species have been recorded on or within sight of the island (B. Walter pers. comm.). Of these, 22

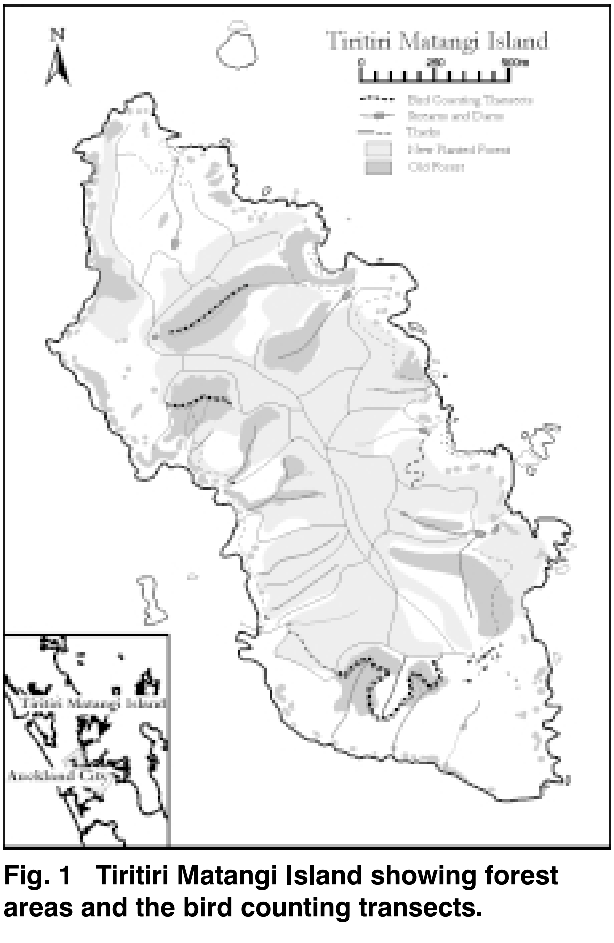

Fig. 1 Tiritiri Matangi Island showing forest areas and the bird counting transects.

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are species which have been introduced to New Zealand and have found their own way to the island; nine are native birds which have been translocated to the island; 19 are native birds which are not known to breed on the island and 27 are native birds which are naturally present and breeding on the island.

This paper reviews the changes in numbers of selected bird species in forest areas which have not been deliberately modified by human activity over the period 1990 to 1998.

METHODS

In 1987 the Ornithological Society of New Zealand (OSNZ) established bird counting transects on Tiritiri Matangi. Data from three transects located in unmanaged forest areas are used in this paper (Fig. 1). The transects were counted in November each year, as close as possible to the 20th of each month. Counters were given the following instruction for counting: "Walk slowly along the transect. Try to keep walking but you may stop to identify a bird. Count each bird within 10 metres of each side of the transect. Record the start and finish time of each count to the nearest minute. Record the average weather experienced during each count. Birds may be recorded as 'seen' or 'heard'". A standard form, with all bird names already entered, was used.

The counts were repeated on two mornings, with a total of six to 10 counts (depending on the number of people participating) being recorded each year. The data shown here are the average number of birds seen and heard for each species for the six to 10 counts on each transect each year.

Two transects were through old pohutukawa (*Metrosideros excelsa*) forest, which at the start of the study period had little understorey, but now has moderately dense understorey. The third transect was through forest dominated by introduced wattle (*Acacia decurrens*) in which native understorey species have increased in variety and density during the study period. There has been no management of these forest areas by humans. A small section of the third transect passed through an area that was grassland at the start of the study and is now planted with native trees.

The planted areas adjacent to these forest areas (Fig. 1) were either predominantly grassland or bracken fern (*Pteridium esculentum*) at the start of the study period, but are now closely planted with native trees varying in height from one to four metres.

Data from the November (austral spring) counts should reflect the resident population, rather than be influenced by the varying abundance of young of the year. We considered the average count recorded for each species, then selected for detailed analysis the five native forest dwelling species which were present before conservation management of the island began, and that have been recorded in the November counts every year. Two of the native species that were re-introduced to the island before these counts began, and two exotic species which are commonly found in forest areas, are also considered.

Analysis

The bird count data (Table 1) were analysed for significant differences between the 1990-1992 and 1996-1998 periods, using 't' tests.

RESULTS

The native species recorded were (in decreasing order of abundance, post eradication): tui (*Prosthemadera novaeseelandiae*), bellbird (korimako) (*Anthornis melanura*), fantail (piwakawaka) (*Rhipidura fuliginosa*), grey warbler (riroriro) (*Gerygone igata*), silvereye (tauhou) (*Zosterops lateralis*), spotless crake (puweto) (*Porzana tabuensis*), kingfisher (kotare) (*Halcyon sancta*), pigeon (kereru) (*Hemiphaga novaeseelandiae*), kaka (*Nestor meridionalis*), long-tailed cuckoo (koekoea) (*Eudynamys taitensis*) and shining cuckoo (pipiwharauroa) (*Chrysococcyx lucidus*),

Seven native species were introduced to the island before or during the study period, so numbers of introduced native species were expected to change. Six of these introduced native species encountered on the three forest transects during the study were (in decreasing order of abundance, post eradication): saddleback (tieke) (*Philesturnus carunculatus*), whitehead (popokatea) (*Mohoua albicilla*), stitchbird (hihi) (*Notiomystis cincta*), red-crowned parakeet (kakariki) (*Cyanoramphus*

Table 1 A comparison of average numbers of birds counted on the three "forest" transects during the three years preceding the rat eradication and three years after rat eradication, with a three-year period between to allow bird species to establish a new equilibrium in the absence of rats.

	1990-1992		1996-1998		% change
Native birds present during all counts:					
Bellbird	8.6	±3.6	16.3	±1.2	90.6
Fantail	3.6	±2.0	3.2	±1.1	-9.7
Silvereye	3.2	±2.2	0.6	±0.5	-81.8
Tui	27.4	±3.2	30.0	±4.6	9.5
Grey Warbler	2.0	±0.2	1.0	±0.3	-48.3
Native birds introduced before counts began:					
Parakeet	2.1	±1.5	5.8	±1.8	178.6
Saddleback	9.9	±1.8	19.0	±2.1	91.9
Exotic birds in the forest:					
Chaffinch	1.0	±0.4	0.7	±0.3	-36.8
Blackbird	1.7	±0.1	1.8	±0.7	5.5

novaezelandiae), robin (toutouwai) (*Petroica australis*), and kokako (*Callaeas cinerea*).

The exotic species recorded were (in decreasing order of abundance, post eradication): brown quail (*Synoicus ypsilophorus*), blackbird (*Turdus merula*), chaffinch (*Fringilla coelebs*), starling (*Sturnus vulgaris*), song thrush (*T. philomelos*), greenfinch (*Carduelis chloris*) and dunnock (*Prunella modularis*).

While trends are apparent, it can be expected that not all species will change in number at similar rates, and some changes may not be related to rat eradication. Five native species, two introduced native species and two exotic species which were present in the forest throughout the study period, and that were recorded during all counts, are examined below in more detail.

Of the native birds that were present during all counts, the 90.6% increase in bellbird numbers is significant (P=0.024) but the 9.5% increase in tui numbers is not statistically significant (P=0.459). Fantail decreased slightly in abundance but this change is not statistically significant (P=0.794). The apparently-large decrease in abundance of silvereye (-81.8%) is not statistically significant (P=0.113) but the lesser decrease in grey warbler (-48.3%) is statistically significant (P=0.015).

Two native bird species were introduced to the island before this study began: parakeets in 1973 and saddlebacks in 1984. Following rat eradication, parakeets increased by 178.6% but this is marginally insignificant (P=0.051). Numbers in the bush transects remain low with high variation from year to year. When introduced to other locations saddlebacks have reached carrying capacity by the sixth year after liberation (Lovegrove pers. comm., Veitch pers. obs.) and so, on Tiritiri Matangi, they may have reached carrying capacity in 1990. Following rat eradication their numbers increased by 91%, which is highly significant (P=0.005).

Two exotic species were recorded in all or nearly all counts: chaffinches and blackbirds. Over the study period the average numbers of both species have declined. The 36.7% decline of chaffinches is not significant (P=0.286), nor is the 5.5% increase of blackbirds (P=0.818). Total numbers of both species were low throughout the study period.

DISCUSSION

Pacific rats are known to suppress the abundance of native fauna (Atkinson and Moller 1990). Their presence on Tiritiri Matangi may have affected the birds counted in this study in three ways: directly through predation; indirectly through competition for food; indirectly through habitat modification. Endeavouring to separate the impacts of the rats from the less-direct impacts caused by the conservation management work on the island is also difficult.

We have endeavoured to exclude the impacts of conservation management work, other than rat eradication, by using data only from the least-modified habitats. However, the planting of trees adjacent to these relatively-small forest areas may have changed bird numbers within the forest. Direct predation by rats is most likely to affect birds such as saddlebacks and parakeets which nest or roost close to, on or in the ground, or in tree cavities accessible to rats. Competition for food may affect all species that feed on fruit, seeds, or large invertebrates at sites accessible to rats. Since rat eradication there has been a massive increase in abundance of ripe fruits and seeds which were previously eaten by rats before they ripened (Veitch pers. obs.). The forest understorey has become notably more dense, which may provide food for some birds or make the forest too dark for others. C. J. Green (pers. comm.) has shown that terrestrial insects have increased significantly since rat eradication; these insects are a principal food for saddlebacks and blackbirds.

Counting forest birds is an imprecise science. The data shown here is very typical of such counts (e.g. Girardet *et al.* 2001) with variability possibly caused by conspicuousness of species, weather patterns, feeding locations, observer aptitude and other factors. This variability makes year to year comparisons impossible, and the comparison of less than three-year periods to be undesirable. In this study we have compared three-year periods and found some changes to be highly significant, with most of these probably resulting from the eradication of rats.

The increase of bellbirds may be a direct result of increased food. Seasonally the fruit on low-growing shrubs is a significant part of the bellbird diet. Their nests are also accessible to rat predation.

The decrease of silvereyes and grey warblers may be a result of changing forest composition. A similar decline of these species has been noted elsewhere (Diamond & Veitch 1981) following regeneration of forests and forest understorey.

Parakeets depend on fruits and seeds as their major food source. Much of their nesting and roosting on Tiritiri Matangi is in crevices in cliffs as few tree cavities are available. This species can survive predation by Pacific rats, but there is a strong indication here that their numbers have increased dramatically as a result of rat eradication.

Saddlebacks on Tiritiri Matangi have been provided with nest and roost boxes and so are mostly safe from predation, apart from juveniles which are likely to spend the first few nights after fledging on the ground. A major saddleback food source is terrestrial insects, and so this species was expected to benefit from rat eradication.

Chaffinches have previously been reported to decline in areas where forest regeneration has occurred (Diamond and Veitch 1981). A similar decline of blackbird numbers was expected but this species may be gaining benefit from improved food in nearby open areas.

We consider that the eradication of rats from Tiritiri Matangi has been a significant factor in the changes of bird numbers recorded here. Some species have benefited in the short term, others may benefit more in the longer term and some species have declined as forest composition changes.

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Spartina anglica eradication and inter-tidal recovery in Northern Ireland estuaries.

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Abstract In 1998 an experiment was initiated to study the effectiveness of eradication methods on two *S. anglica* swards. The effects of eradication treatments on live *S. anglica* stem density and other associated plant species were examined using the herbicides glyphosate and Dalapon, smothering with black plastic sheeting, and cutting. Glyphosate was relatively ineffective. Dalapon applied at a rate of 57 kg/ha and smothering were the most effective methods, reducing live *S. anglica* stem density by over 95% within one year. *S. anglica* re-establishment occurred over the two years following treatment applications indicating that eradication would require re-application of treatments. Cutting treatments in this study increased the abundance of *Puccinellia maritima* within one of the swards, suggesting that it may facilitate the establishment of other saltmarsh species. Legal constraints and limitations of resources makes eradication of *S. anglica* in most Northern Ireland estuaries unlikely. It may be possible to contain the current spread of *S. anglica* by removing seedlings, clumps and tussocks, whilst attempting to convert sward areas into mixed saltmarsh.

Keywords Control; mudflats; *Puccinellia*; saltmarsh.

INTRODUCTION

Spartina anglica C. E. Hubbard originated at Hythe, Southampton Water, England, in the nineteenth century (Gray *et al*. 1991). *Spartina anglica* was the result of chromosome doubling by *Spartina x townsendii* H. and J. Groves, the sterile hybrid between the native European *Spartina maritima* (Curtis) Fernald and the introduced North American *Spartina alterniflora* Loisel (Gray *et al.* 1991). *S. anglica* has a relatively narrow ecological amplitude. Gray et al. (1995) state that "broadly speaking *Spartina anglica* is distributed between Mean High Water Neap tides (MHWN), and Mean High Water Spring tides (MHWS)" in south and west Britain. This comprises a range of low-high elevation estuarine habitat.

As *Spartina* spp. grow they can accrete large volumes of tidal sediment leading to substantial increases in marsh elevation. This property made *Spartina anglica* a valuable species for coastal protection and reclamation schemes in the early twentieth century (Ranwell 1967). *S. anglica* was planted in Northern Ireland estuaries during the 1920-1950s (Bleakley 1979) and is currently expanding its range.

S. anglica spread occurs in two phases, initial invasion and establishment of seedlings or plant fragments on open mudflats, and then expansion of tussocks by radial clonal growth. Spreading tussocks fuse to form clumps that can expand into extensive meadows. Seed production of *S. anglica* is variable both temporally and spatially (Gray *et al.* 1991). It is thought that *S. anglica* does not form a seedbank in estuarine substrates.

Several Northern Ireland estuaries are of international importance for wildfowl and waders, such as an over-wintering population of pale-bellied brent geese (*Branta bernicla hrota*). Both estuaries in this study, Strangford Lough and Lough Foyle, have been designated as 'Ramsar' sites. The introduction and spread of *S. anglica* into wildfowl and wader feeding areas is seen as a threat to bird populations. *Zostera* spp. beds, which are an important food source for wildfowl in Northern Ireland, may decline in abundance due to *S. anglica* invasion (Oliver 1925; Madden *et al.* 1993). Waders are also likely to be affected by *S. anglica* invasion as dense stands physically prevent their access to invertebrate prey species inhabiting the sediments of *S. anglica* swards.

Since the late 1960s attempts have been made to control and eradicate *S. anglica* in Northern Ireland. Dalapon (2,2 dichloropropionic acid) application has been the main method used, but digging was also attempted. Digging was, however, only successful on plants smaller than 50 cm in diameter (Furphy 1970). Early trials in Britain suggested that Dalapon was one of the most effective herbicides for eradicating *S. anglica*, achieving over 90% kill (Ranwell and Downing 1960; Taylor and Burrows 1968). Dalapon is, however, no longer manufactured and the Environment and Heritage Service in Northern Ireland, which is responsible for management of Northern Ireland estuaries, requires a replacement herbicide for *S. anglica* eradication. Several other herbicides have been tried in *Spartina* spp. eradication experiments in other countries. Of these, fluazifop-P-butyl, haloxyfop, and imazapyr have achieved over 90% *Spartina* spp. kill (Pritchard 1996; Shaw and Gosling 1996). Glyphosate, however, is to date, the only other herbicide licensed for use in estuarine environments in Northern Ireland. Licensing of other herbicides is likely to be a costly and slow process. The greatest successes using glyphosate, achieving over 75% kill, have been obtained using glyphosate along with an added surfactant (Garnett *et al.* 1992; Kilbride *et al*. 1995; Crockett 1997, Major and Grue 1997; Norman and Patten 1997). Surfactants are currently banned from use in Northern Ireland inter-tidal areas. Previous work suggests that applications of glyphosate on its own produces poor *S. anglica* kill rates (Garnett et al. 1992).

Pages 124-131 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

The Environmental and Heritage Service also wanted to investigate the potential of non-herbicidal methods for *S. anglica* eradication due to environmental and health concerns about herbicide use, and due to a ban of herbicide use in shellfish designated areas (see Discussion). Smothering and burying are the only non-herbicide techniques that have reduced *Spartina* spp. stem density by over 90%. Initial attempts at burying using a rotoburying machine at Lindisfarne (England) resulted in over 95% *S. anglica* kill (Davey *et al*. 1996). Rotoburying machine use is unsuitable in Northern Ireland estuaries due to soft sediments. Smothering is therefore a more suitable option. Covering plants with black plastic sheeting prevents photosynthesis, and probably leads to increases in the temperature of sediments, thus leading to plant death. American and Australian studies using black plastic to smother *Spartina* spp. have reported kill rates of up to 99-100% (Aberle 1990; Lane 1996).

This study assesses the effectiveness of Dalapon and glyphosate for eradicating *S. anglica* in two swards. Cutting prior to herbicide application was also examined to determine if it increased *S. anglica* kill rates. These methods were compared with the non-herbicidal eradication method, smothering with black plastic sheeting. In addition the previously-unexamined effects of eradication treatments on other plant species within *S. anglica* swards were investigated.

METHODS

Study area

Two sites were selected for the *S. anglica* eradication trials. These sites were the only areas available for this study due to a herbicide ban in other locations. One of the sites was a 1.4 ha *S. anglica* sward at Lough Foyle (Fig. 1, 2; Grid ref. 55º 03'N, 7º 02.6'E) and the other a 0.15 ha *S. anglica* sward at Strangford Lough (Fig. 1, 3; Grid ref. 54º 31.8'N, 5º 40.6'E). Relatively uniform areas of *S. anglica* were selected for placement of experimental plots to avoid gullies.

Lough Foyle is a 200 km^2 marine inlet at the northern coast of Northern Ireland, with a tidal range of 1–2 m. *Spartina* spp. were introduced into the sheltered bay containing the trial plots in the 1930s. The plots within the *S. anglica* sward receive no tidal inundation at MHWN tides. During MHWS tides inundation levels range between 20-32 cm. *S. anglica* within the study plots had a mean stem density of 232 stems per square metre, and a mean stem height of 33.2 cm in July 1998. During this trial *S. anglica* was ob-

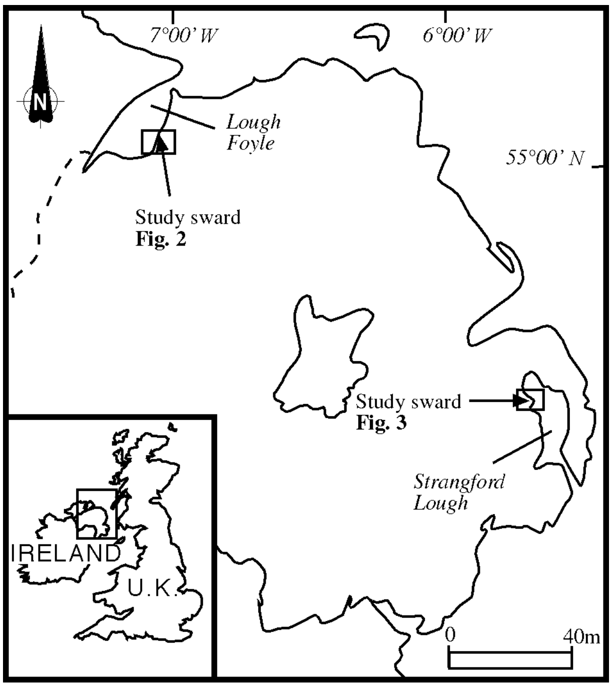

Fig. 1 Location of study areas in Northern Ireland.

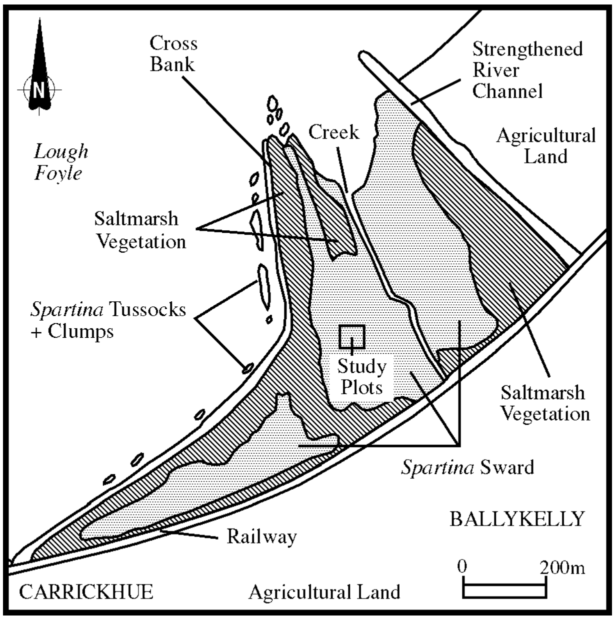

Fig. 2 *Spartina anglica* study sward, Lough Foyle.

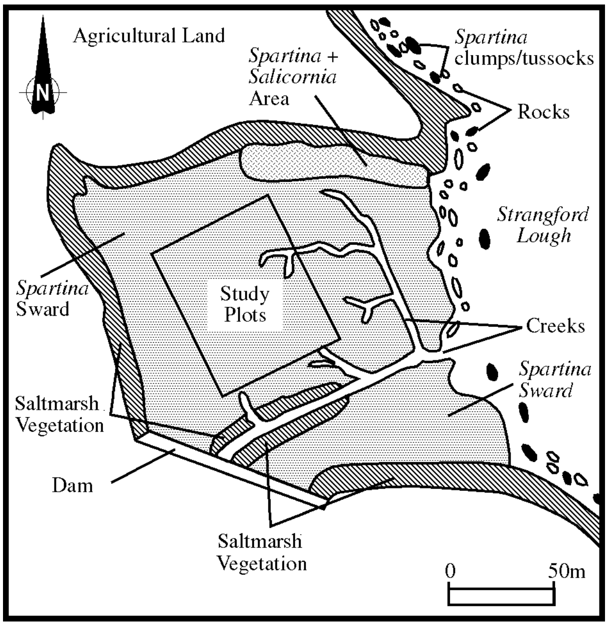

Fig. 3 *Spartina anglica* study sward, Strangford Lough.

served to begin growth in April and began flowering in June-July. The study area contained *Puccinellia maritima* (median Domin value 3-4), *Aster tripolium* and *Plantago maritima* (individuals of both, median Domin value <1) prior to the study. The vegetation community within the sward is similar to those found in mid-elevation *S. anglica* swards in other U.K. and Netherlands estuaries (cf. Brereton 1971; Adam 1981; Roozen and Westhoff 1985; Scholten and Rozema 1990; Gray 1992). Several saltmarsh strips dominated by *Puccinellia maritima* (Domin scale 6), *Agropyron pungens, Aster tripolium, Plantago maritima*, *Festuca rubra* (all Domin scale 5), and *Triglochin maritima* (Domin scale 4), with individuals of several other saltmarsh species including *Chenopodium rubrum,* were close to the sward (Fig 2).

Strangford Lough is a 150 km^2 marine inlet on the south-eastern side of Northern Ireland, with a tidal range of 2-3.5 m. *Spartina* spp. were introduced into Strangford Lough in the 1930s and 1940s. The *S. anglica* sward in this study was first recorded in 1969 and is confined to a sheltered bay. The plots in the sward receive no tidal inundation at Mean High Water Neap tides. During Mean High Water Spring tides inundation levels range between 51-67 cm. *S. anglica* within the study plots had a mean stem density of 336 stems per square metre, and a mean stem height of 23.7 cm in July 1998. During the study *S. anglica* was noted to begin growth in April and began flowering in June-July. *Puccinellia maritima* individuals were recorded (Domin value 1) in the experimental plots prior to the study. *P. maritima* and *Aster tripolium* occurred in raised micro-hummocks adjacent to the experimental plots, and *Salicornia* spp. were observed in other adjacent sward areas. The vegetation communities within the sward are similar to those found in low-elevation *S. anglica* swards in other U.K. and Netherlands estuaries (cf. Brereton 1971; Adam 1981; Roozen and Westhoff 1985; Scholten and Rozema 1990; Gray 1992). Several saltmarsh strips dominated by *Puccinellia maritima* (Domin scale 6), *Aster tripolium*, and *Plantago maritima* (both Domin scale 4), with a lower abundance of several other saltmarsh species, were close the sward (Fig. 3). An area dominated by *Spartina anglica* and *Salicornia* spp. (both Domin scale 8), was close to the study plots (Fig. 3).

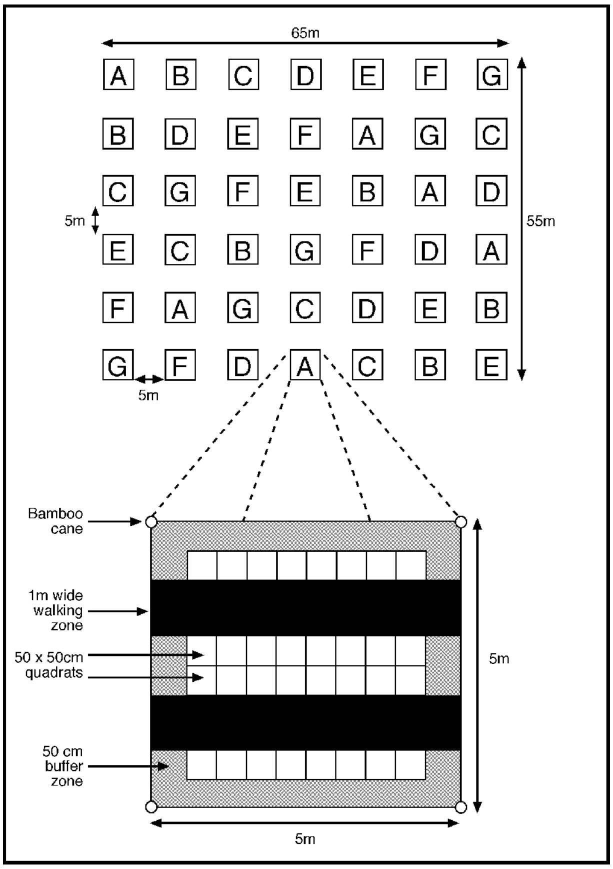

Fig. 4 Example of random block layout and experimental plot design. Treatments: A = Dalapon; B = Sward cut and glyphosate; C = Experimental Control; D = Glyphosate; E = Sward cut and smother; F = Sward cut; G = Sward cut and Dalapon.

Experimental layout

Six replicate plots were used, with seven different treatments. Plots of 5 m x 5 m were laid out in a random block formation angled approximately parallel to the shoreline, with a separating distance between plots of 5 m (Fig. 4). Within each plot, two 1m walking strips were retained for access when applying treatments and monitoring. A buffer zone of 50 cm was established around the inner edge of the plot. This area was not used for recording. The remaining areas were divided into thirty-two 0.5 m x 0.5 m quadrats for experimental recording.

The seven treatments applied were :

- Experimental Control (no treatment)
- Dalapon applied at a rate of 57 kg/ha
- Glyphosate without added surfactant applied at a rate of 5.0 *l*/ha
- Sward cut to 10 cm
- Sward cut to 10 cm and Dalapon applied after six weeks growth
- Sward cut to 10 cm and glyphosate applied after six weeks growth
- Sward cut to 10 cm and covered with black plastic sheeting for six months

Treatment application

The Dalapon application rate was suggested by Ranwell and Downing (1960) and Taylor and Burrows (1968). The form of Dalapon available for use was Farmon Dowpon, a wettable powder containing 85% of the sodium salt of Dalapon. The glyphosate application rate used is recommended by Monsanto to control grasses in the aquatic environment, and has previously been used by Garnett *et al.* (1992). The form of glyphosate used was Roundup Biactive, an aqueous concentrate containing 360 g/*l* glyphosate acid present as 480 g/*l* of the isopropylamine salt of glyphosate. Herbicides were applied using a Cooper Pegler CP15 knapsack sprayer. The sprayer was operated at a pressure of 1 bar (15 psi) and was fitted with a red floodjet/deflector nozzle that had a 2.0 m spray width from a nozzle height of 50 cm above the target. Herbicides were applied at least six hours before tidal inundation during neap tides, on cloudy, rainless days with wind speeds of less than 10 km/hr. Spraying was carried out in August 1998, before *S. anglica* seedheads had developed.

Mowing was accomplished with a hand-held brush cutter during July 1998. Cutting was done to within 10 cm of the substrate. Cut material was raked to one side and subsequently removed by tides. Follow-up herbicide applications were carried out six weeks later in August.

Industrial strength black plastic sheeting was cut into appropriately sized strips and laid out onto the plots during July 1998. Galvanised wire mesh was laid on top of the sheeting, extending beyond the edge of the plastic. Galvanised steel wire pegs were staked through the plastic and wire mesh to hold both layers in place. The plastic sheeting was removed in January 1999.

Records and analysis

The first data collection was carried out during July 1998, prior to the application of treatments. Recording was repeated in July 1999 and July 2000.

Every plot contained thirty-two 50 cm x 50 cm quadrats. Five randomly-drawn quadrats per plot were used to record live *S. anglica* stem density and Domin values of other plant species. Different quadrats were selected for each years recording. The number of live *S. anglica* stems in each quadrat were counted. The mean of the five stem density counts per plot was used to represent stem density of the plot, thus avoiding sacrificial pseudoreplication. The same five quadrats were used in each plot to estimate percentage cover of all plant species present excluding *S. anglica*. The mean percentage cover value from the five quadrats was calculated and converted into a Domin value for each species per plot.

For each year's results, Kruskal-Wallis tests were used to analyse differences between live *S. anglica* stem densities, and the abundance of saltmarsh plants, in the seven treatment groups (Sokal and Rohlf 1998). All statistical analysis was carried out using the statistical computer package SPSS Version 9.

RESULTS

Live *S. anglica* stem density

There was no significant difference between the mean live *S. anglica* stem density of the seven treatment groups prior to treatment application at both sites in July 1998. Significant differences (P, <0.001) amongst the live *S. anglica* stem densities of the treatment groups at each site were observed in July 1999 and July 2000 (Table 1).

The Dalapon, Cut + Dalapon, and Cut + Smothered treatments caused over 95% reductions in live *S. anglica* stem density at the Lough Foyle site between July 1998 and July 1999 (Table 1). The Cut, and Cut + Glyphosate treatments resulted in increases in stem density whilst the Experimental Control and Glyphosate treatments experienced reductions in stem density. By July 2000 the Cut + Smother treatments achieved over 90% reductions of live *S. anglica* stem density compared with pre-treatment levels. The live stem density levels within the Cut + Dalapon, and Dalapon treatments had increased between July 1999 and July 2000 resulting in approximately 60% reductions compared to pre-treatment levels. Stem densities also increased in the Experimental Control and Glyphosate treatments between July 1999 and July 2000. During the same time period the Cut + Glyphosate, and Cut treatments experienced reductions in live stem densities.

Table 1 Percentage changes in live *S. anglica* stem density between July 1998 (pre-treatment) and July 1999, and from July 1998 to July 2000 at Lough Foyle and Strangford Lough.

	Lough Foyle		Strangford Lough	
Treatment	1998 - 1999	1998 - 2000	1998 - 1999	1998 - 2000
Cut	+ 50.3	+ 1.7	- 2.0	+ 2.0
Cut + Dalapon	- 96.8	- 58.1	- 99.6	- 98.0
Cut + Glyphosate	+ 58.8	+ 10.6	- 25.4	- 11.3
Cut + Smother	- 98.9	- 90.3	- 99.9	- 99.9
Dalapon	- 96.3	- 57.5	- 95.8	- 92.3
Glyphosate	- 14.8	+ 69.1	- 52.2	- 30.0
Experimental Control	- 15.3	+ 45.3	- 52.6	- 50.1
Significant difference between groups	< 0.001***	< 0.001***	< 0.001***	< 0.001***

Significance level: *P<0.05, **P<0.01, ***P<0.001

The Dalapon, Cut + Dalapon, and Cut + Smother treatments caused over 95% reductions in live *S. anglica* stem density at the Strangford Lough site between July 1998 and July 1999 (Table 1). During the same time period the Experimental Control and Glyphosate treatments experienced approximately 50% reductions of live stem density, the Cut + Glyphosate treatment reductions of 25%, and the Cut treatment a reduction of 2%. By July 2000 the Cut + Smother, and Cut + Dalapon still had live *S. anglica* stem density reductions of over 95% compared to pre-treatment levels, whilst the Dalapon treatment had reductions of over 90%. The Experimental Control had similar live stem densities as July 1999 levels. The Glyphosate, Cut, and Cut + Glyphosate treatments all experienced an increase in live stem density between July 1999 and July 2000.

Abundance of other plant species

In the Lough Foyle sward individuals of *Aster tripolium*, *Plantago maritima*, and *Puccinellia maritima* were recorded in plots during July 1998, July 1999, and July 2000. An individual of *Chenopodium rubrum* was recorded in July 2000. There were no significant differences in the abundance of *A. tripolium*, *Plantago maritima*, and *C. rubrum* between treatment groups in any year. There were no significant differences in *Puccinellia maritima* abundance between the seven different treatment groups in July 1998 prior to treatment application (Table 2). In July 1999 (one year after treatment application), there was a significant difference between the treatments. Since July 1998 the largest reductions in median *P. maritima* abundance occurred in the Dalapon, Glyphosate, and the Cut + Smother treatments. In the Experimental Control plots, and the Cut + Dalapon plots, smaller reductions in median *P. maritima* abundance were noted. The only treatments in which median *P. maritima* abundance increased were the Cut + Glyphosate, and Cut treatments. In July 2000 (two years after treatment application), there was no significant

Table 2 Median *Puccinellia maritima* Domin scale abundance in Lough Foyle plots in July 1998, July 1999 and July 2000.

Treatment	*Puccinellia maritima* abundance		
	1998	1999	2000
Cut	3.5	5.5	4.5
Cut + Dalapon	1	0.5	1.5
Cut + Glyphosate	3	6	6.5
Cut + Smother	4.5	1	4
Dalapon	3	0	3
Glyphosate	5	2	3.5
Experimental Control	1.5	1	1
Significant difference between groups	0.340	0.013*	0.247

Significance level: *P<0.05, **P<0.01, ***P<0.001

difference in *P. maritima* abundance between the treatment groups. Most of the treatment groups experienced an increase in *P. maritima* abundance between July 1999 and July 2000, except the Experimental Control which remained similar to July 1999 levels and the Cut treatment which showed a decline in *P. maritima* abundance.

In the Strangford Lough sward *Puccinellia maritima* was recorded in only one experimental plot in July 1998, July 1999, and July 2000. There were no significant differences between *P. maritima* abundance between treatment groups in any year. No other species apart from *S. anglica* and *P. maritima* were recorded in the Strangford Lough plots during July 1998 or July 1999. In July 2000 low abundance values (maximum 1%-4% cover) of *Salicornia* spp. were recorded in the Dalapon, Cut + Dalapon, and Cut + Smother plots. There were no significant differences between *Salicornia* spp. abundance and treatment groups.

DISCUSSION

S. anglica eradication techniques

Herbicides are the most frequently used *Spartina* spp. control method due to their practical ease of use and cost-effectiveness. This study shows that when used in suitable conditions, Dalapon applied at a rate of 57 kg/ha will cause over 95% reduction in live *S. anglica* stem density within the first year. Glyphosate was as ineffective with similar live *S. anglica* stem densities as the Experimental Control plots after one year. Cutting had no additive effect when applied before Dalapon application in this experiment. Preliminary results of an experiment in Washington, U.S.A, using a single cut of *S. alterniflora*, followed by glyphosate application gave a similar outcome (Major and Grue 1997). The single Cut treatments produced the highest live stem density values at each site in this study. A single cut will therefore not assist with *S. anglica* eradication. At Lough Foyle, live *S. anglica* stem density in Cut plots was lower than the Experimental Control after two years. This may indicate that rhizome energy reserves were extensively used-up in the year following the cut. It has been suggested that multiple cutting may reduce *S. anglica* vigour and reduce above ground biomass (Scott *et al*. 1990), but it is also possible that certain cutting regimes would cause increases in stem density (Hubbard 1970). Smothering caused over 95% reductions in live *S. anglica* stem density within the first year of application.

The experimental treatments failed to achieve 100% kill of *S. anglica*. Eradication would require repeat applications of eradication treatments, possibly on many occasions. In this study *S. anglica* re-establishment was more rapid in the mid-elevation Lough Foyle sward compared to the low-elevation Strangford Lough sward. Ranwell and Downing (1960) reported the complete recovery of *S. anglica* within sprayed areas two years after Dalapon application, whilst Taylor and Burrows (1968) reported 88%-98% reductions in *S. anglica* stem density two years after Dalapon application. This suggests that site specific factors will influence *S. anglica* re-establishment rates after treatment applications. The rapid recovery of *S. anglica* in some sites suggests that treatment re-application should occur in the year following initial application of treatments. Site-specific factors are also probably responsible for the unexplained difference in live *S. anglica* stem density in the Experimental Control plots between study sites.

Effects of *S. anglica* eradication treatments on other saltmarsh species

Any plant species in the *S. anglica* dominated sward that is within the range of plants affected by an applied herbicide, is likely to be killed. Reductions in abundance of *Salicornia* sp., *Suaeda* sp., and *Puccinellia* sp. have been noted when glyphosate is applied to *S. anglica* swards (Garnett *et al.* 1992). In the present study both Dalapon and glyphosate caused reductions in *Puccinellia maritima* abundance at Lough Foyle. Smothering should kill all vegetation due to the exclusion of light. *P. maritima*, and *Salicornia* spp. have been killed by algal mats and tidal litter due to the effect of smothering (Jefferies *et al.* 1981; Langlois *et al.* 2001). *P. maritima* abundance declined in smothered plots at Lough Foyle between July 1998 and July 1999. During the second year of the study *P. maritima* was noted to increase in abundance at Lough Foyle in the Dalapon and Smothered plots. This indicates that substrate and environmental conditions remaining after *S. anglica* removal are suitable for colonisation by other species.

Cutting or grazing of *S. anglica* swards may promote the growth of other species such as *P. maritima* (Beeftink 1985; Scholten and Rozema 1990; Scott *et al.* 1990). *P. maritima* abundance increased at Lough Foyle in the Cut + Glyphosate, and Cut plots over the first year of this experiment. The reduction of *S. anglica* height caused by cutting allows increased light penetration within the canopy, thus improving the growth of other light dependent species (Scholten and Rozema 1990). There was no further increase in *P. maritima* abundance in the Cut, and Cut + Glyphosate plots during the second year of this investigation. This suggests that the opportunity for *P. maritima* spread was short-lived. *S. anglica* growth during the growing season would have increased the height of the *S. anglica* canopy, reducing light penetration and thus hampering further spread of the lower-lying *P. maritima*.

In this study colonisation of other species was at a low level, suggesting that seed input into treated areas from surrounding saltmarsh vegetation is low. Other studies suggest that saltmarsh species, such as *Salicornia* spp., form no long-term seedbank in substrates and that the majority of seeds of many saltmarsh species fall within centimetres of the parent plant (Jefferies *et al.* 1981; Gray and Scott 1977; Ellison 1987; Hartman 1988). The species that colonise will be dependent upon local environmental conditions in relation to the regeneration niche of the individual species (Beeftink 1985) and the abundance of adult plants of each species in surrounding areas (Rand 2000). In this study the elevation of the two swards influenced the colonising species. The low-elevation Strangford Lough was suitable for *Salicornia* spp. colonisation, whilst the mid-elevation sward at Lough Foyle was suitable for colonisation by *Aster tripolium*, *Plantago maritima*, and *Puccinellia maritima*. The persistence of any colonising species will be affected by its competitive ability against other saltmarsh species, especially in areas were *S. anglica* re-establishment after control is rapid. *Puccinellia maritima*, for example, will outcompete *S. anglica* in northern latitudes (in the northern hemisphere) in upper marsh elevations with sandy nutrient-rich sediments (Scholten and Rozema 1990; Huckle *et al.* 2000).

Considerations for *S. anglica* management in Northern Ireland estuaries

The 95% reduction in live stem density caused by Dalapon applications or smothering treatments in this study suggests that eradication of *S. anglica* is feasible if treatment applications are repeated. We advise that treatment re-application begins in the year following initial applications as *S. anglica* recovery can be rapid. Treatments may have to be repeated on several occasions to achieve successful eradication. Smothering has proven to be effective, but the practicalities, cost-effectiveness and environmental impacts of using large-scale smothering are untested. Herbicides are the most cost-effective and practical eradication methods, but glyphosate is not a suitable replacement for Dalapon. Attempts should therefore be made to find a suitable herbicide replacement for use in Northern Ireland estuaries, possibly by obtaining off-label permits. This process would take a number of years as toxicity studies, risk assessments, and cost/benefit analyses are required before the herbicide is permitted for use in Northern Ireland estuaries. Research from other countries suggests that the herbicides fluazifop-P-butyl, haloxyfop, and imazapyr are worthy of further investigation (Pritchard 1996; Shaw and Gosling 1996).

Several issues are likely to constrain the effectiveness of eradication attempts in Northern Ireland, such as limitations of economic resources, the abundance of *S. anglica* within estuarine systems, public objections, and legal restraints (cf. Kriwoken and Hedge 2000). In Northern Ireland there is currently a ban on the use of herbicides in shellfish designated areas; the result of a legal dispute that occurred after an experiment to eradicate *S. anglica* in 1980 (Kirby 1994). A local oyster farmer settled out of court after claiming that the removal of *S. anglica* resulted in the liberation of silt, which subsequently smothered and killed his oysters. In these areas only minimal herbicide application is permitted. It is therefore unlikely that *S. anglica* will be eradicated from Northern Ireland estuaries in the near future. An alternative management strategy of eradication from selected estuaries/areas and containment is needed.

Areas with no legal restraints against herbicide use, and areas with high environmental value such as wildlife reserves or heavily-used recreation areas could be targeted for eradication using herbicide. We suggest that an initial phase of eradication could focus on preventing *S. anglica* establishment into new areas, eradication of tussocks and clumps, and preventing expansion of sward areas (see Moody and Mack 1988). Once achieved, annual monitoring and removal of *S. anglica* seedlings is required to keep these areas free from *S. anglica* re-establishment. The possible long-term cost of this proposal should be considered in any future management scheme.

The next phase of eradication could focus on low-elevation *S. anglica* swards. These swards will be more prone to erosion after *S. anglica* removal than mid-high-elevation swards. Eradicated areas may erode to former mudflat levels within three years (McGrorty and Goss-Custard 1987), or be colonised with low-marsh vascular species such as *Salicornia* species. This is likely to result in mud flat that is suitable for use as feeding grounds for wildfowl and waders (McGrorty & Goss-Custard 1987). Continuous monitoring and removal of *S. anglica* seedlings would be required in these areas if *S. anglica* plants remain within the estuarine system. The next phase of eradication could focus on mid-high elevation swards. High-level marsh may require a period of up to 20 years to erode to low-elevation mudflat after *S. anglica* eradication (McGrorty and Goss-Custard 1987). During this time the area would be open to colonisation by other saltmarsh species and develop into saltmarsh, rather than mudflat.

In areas where *S. anglica* eradication is not feasible containment strategies are suggested. Initial attention could focus on preventing further spread of *S. anglica*, especially into sites of environmental importance. Herbicides can be used in any area where they are permitted. Smothering may be suitable for killing small-scale *S. anglica* infestations in areas where herbicide use is banned, but will probably be unsuitable for large sward areas. It may also be possible to use control techniques, such as cutting, to encourage colonisation by, and growth of, other saltmarsh species within *S. anglica* swards, in order to promote the development of a mixed saltmarsh community. *Salicornia* spp. are the most likely colonisers of low-elevation sites. If *S. anglica* regrows it will outcompete *Salicornia* spp. (Beeftink 1985; Ellison 1987), and this could result in the area returning to mono-dominant *S. anglica* sward.

The lack of seed arriving into the controlled areas is likely to be a major factor in hampering the conversion of *S. anglica* swards into mixed saltmarsh communities (Hartman 1988; Rand 2000). There have been no studies that examine attempts to increase the abundance of native saltmarsh species within *S. anglica* swards. It may, however, be possible to overcome the lack of seed input into the area by using species transplants or seed additions. This alternative management method requires further investigation to evaluate its potential success.

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Eradication of feral goats and pigs and consequences for other biota on Sarigan Island, Commonwealth of the Northern Mariana Islands.

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Abstract Sarigan Island (*c*.500 ha) is one of the 15 Mariana Islands in the tropical western Pacific Ocean. The native forest on Sarigan was in an advanced state of decline due to the presence of feral goats (*Capra hircus*) and pigs (*Sus scrofa*). During January and February 1998, 68 pigs and 904 goats were removed by helicopter shooting, ground shooting, trapping, and tracking with dogs. The goal was to stop and reverse the loss of forest and accompanying erosion and thus improve habitat for the endangered Micronesian megapode (*Megapodius laperouse*) and other native species. Follow-up control in 1999 and 2000 removed an additional six goats. Sarigan Island is now considered free of feral ungulates. Vegetation monitoring before and after eradication shows an increase in plant species richness, an increase in tree seedlings, and the rapid expansion of the introduced vine *Operculina ventricosa*. Skinks also increased, but numbers of fruit bats, land birds, and rats have not yet showed change. It is still undetermined as to what effect the vine *Operculina ventricosa* will have on the regeneration and expansion of the native forest.

Keywords Vegetation; megapode; *Operculina ventricosa.*

INTRODUCTION

Of the 11 islands in the Mariana chain (15 islands total) that are uninhabited, the largest five have feral animals. The uncontrolled existence of these populations jeopardises the continued existence of the unique native plant and wildlife species on these islands. Entire forests are disappearing and ecosystems are being changed before they are even understood. Some form of programme, either periodic control or total eradication, needs to be implemented before irreversible damage is done to the entire system.

Sarigan Island had been cultivated and maintained for copra production in 1900 (Fritz 1902). Feral goats (*Capra hircus*) and pigs (*Sus scrofa*) have been present on Sarigan for at least 50 years according to Mr Yamamoto (pers. comm.), a 1940s resident of the island. All residents were evacuated in 1945. In 1950 a request was made to the U.S. Navy administration to let Mr Palacious and company return to the island and commercially harvest goats (J. Johnson pers. comm.); permission was denied. It is believed that ungulate populations were semi-controlled through sporadic harvesting until the 1970s when the most recent attempt at human colonisation was abandoned. Other visitation/harvesting has consisted of brief stops by fisherman or government scientific trips.

The combination of feral goats and pigs has had a severe impact on the native flora and fauna. It appears that the feral ungulates were changing Sarigan from a tropical forest to a grassland habitat (Ohba 1994; unpub. reports CNMI-DFW 1988-1997). This alteration of habitat was believed to be adversely impacting endangered and resource species such as the Micronesian megapode (*Megapodius laperouse*), Mariana fruit bat (*Pteropus mariannus*), and coconut crab (*Birgus latro*) as well as other native species. The primary goal of this project was to remove the feral ungulates from Sarigan Island as a means to improve habitat (through vegetation recovery) for endangered Micronesian megapodes.

METHODS

Study area

Sarigan Island is a relatively small uninhabited island of about 500 ha located 121 miles north of Saipan in the Commonwealth of the Northern Mariana Islands (CNMI) (16º 42' N 145º 46' E). This island is a volcanic cone with steep slopes and no protected beaches. In the central-northwest portion of the island lies a level plateau, which again rises steeply up the main cone. The south and east sides are extremely steep and rise continually to the top of the island (549 m). The vegetation (Ohba 1994) consists of coconut trees (*Cocos nucifera)* with patches of hibiscus (*Hibiscus tiliaceus*) on the lower slopes of the west and north sides. The upper plateau is half native forest (upland mesic climax) and half short grass (*Chrysopogon aciculatus*). The slopes on the south and upper west sides are swordgrass (*Miscanthus floridulus*). The main cone is swordgrass on the southwest and ferns (*Pteris sp.*) on the northeast, with some remnant native forest scattered in ravines and crevices. There are no streams or free-standing water.

Timeline

Between 1995-1997, CNMI-Division of Fish & Wildlife (DFW) formulated a five-phase plan to eradicate pigs and goats on Sarigan:

- Phase I - Reconnaissance and survey
- Phase II - Base camp establishment
- Phase III - Shooting programme

Pages 132-140 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

- Phase IV - Removal of remnant populations/individuals
- Phase V - Follow-up monitoring and re-surveying

CNMI-DFW established 13 photo vegetation plots and completed a general aerial reconnaissance (including photographs) in mid February 1997. Phase I was completed with the help of a US Geological Survey-Biological Research Division (USGS-BRD) bird survey crew in March 1997 that resulted in baseline data on vegetation, bird, bat, and lizard populations (Fancy *et al.* 1999; unpub. report CNMI-DFW 1997,1998).

Phase II, base camp establishment, was generally completed by July 1997. During this period four 3 m x 2.5 m x 2.5 m weatherproof containers, purchased by DFW, were flown onto the island by U.S. Navy helicopter. DFW personnel then moved equipment and supplies into these containers over a period of several months.

Zoology Unlimited was contracted (for USD180,000) by the United States Fish and Wildlife Service through an agreement with the U.S. Navy and CNMI, to conduct phases III and IV. We began Phase III on 1 January 1998 with helicopter shooting followed by ground hunts. This gradually shifted into Phase IV by the beginning of February. Phase IV ended by 1 March 1998 when it was believed all ungulates had been removed from the island. All personnel had left the island by 2 March 1998.

Phase V, the monitoring phase, began in August 1998. This first trip was conducted by CNMI-DFW. Three additional monitoring trips in January and July 1999 and July 2000 were made (unpub. report Zoology Unlimited for USFWS-Honolulu 1999, 2000). During these monitoring trips data on habitat and wildlife was also collected by various biologists. Monitoring of wildlife populations and surveillance for any surviving goats or pigs is ongoing.

Vegetation and wildlife monitoring

Vegetation

In February 1997, one year before the eradication started, 13 permanent vegetation plots were established. These were marked with steel bars and the Universal Transverse Mercator (UTM) coordinates were located using a Global Positioning System (GPS) (unpub. report Zoology Unlimited for USFWS-Honolulu 2000; unpub. report CNMI-DFW 1997). Photos of the plots were taken which included two 3 m poles spaced five metres apart for scale. All plant species within a 2 m^2 area centred on the rebar were identified. Ground cover within the 2 m^2 area was visually estimated as was canopy cover. Trees were surveyed using the point-quarter method. Plots were spaced 100 metres apart and followed a transect that cut across the centre of the island starting in grassland, continuing through native forest, and ending in the coconut forest. Plots were re-surveyed in August 1998, July 1999, and July 2000 and Laura Arriola of CNMI-DFW did analysis and graphs (unpub. report CNMI-DFW 1998-2000).

Bats

Fruit bats were surveyed by various biologists using different methods. Fancy *et al.* (1997) and Morton *et al.* (2000) used Variable Circular Plot (VCP) counts. Wiles in 1983 and 1999 (Worthington 2001) and Johnson in 2000 (unpub. report CNMI-DFW 2000) used station counts, individual sightings, and colony counts to estimate populations. These different methods have been combined to form an estimate.

Birds

Bird surveys on Sarigan were conducted using standard VCP methodology. Different surveyors used either VCP (Fancy *et al.* 1997) or fixed-radius (Morton *et al.* 2000) analytical techniques to estimate densities. Due to the inherent differences between techniques, only the “birds detected per station” were compared. Surveys were done in September 1990, March 1997, July 1999, and July 2000. Dr Justine de Cruz of CNMI-DFW compiled data and conducted the July 1999 and July 2000 surveys (unpub. report CNMI-DFW 1999, 2000).

Rats

Pacific rats (*Rattus exulans*) were trapped using a large snaptrap baited with peanut butter and set on the ground. Thirty traps, spaced every 25 metres, were set on the ground in two separate transects for a total of 60 traps. Traps were left overnight and checked in the morning. Rat trapping was done in July 1999 and July 2000. Scott Vogt of CNMI-DFW conducted surveys and provided data (pers. comm.).

Lizards

Glueboard traps were used for catching lizards. A line of 12 traps, with 5 m spacings was set in the morning and picked up in the afternoon. Traps were set in the shade in both the coconut forest and native forest. Catch rates were expressed as the number of lizards captured per trapping hour. Scott Vogt of CNMI-DFW conducted surveys and provided data (pers. comm.).

RESULTS

Overall we removed 904 goats (with a sex ratio of 2.38 females to every male of the 383 we sexed), 68 pigs (2.06 females to every male of the 55 we sexed), and two cats in the first 60 days. A further six goats were removed during the four follow-up trips to the island; a density of 1.83 goats/ha and 0.14 pigs/ha. The four follow-up trips were made in August 1998, January 1999, July 1999, and July 2000. Each trip was an average of five search days. These trips accounted for one, four, one, and zero goats respectively (four females and two males). No pigs were ever encountered during follow-up trips.

Table 1 Effort and feral ungulate kills for Sarigan Island, CNMI. 1 January to 28 February 1998.

Operational tactic	Hunting effort (estimated)	Goats killed	Pigs killed
Initial aerial	21 hrs flying	344	0
Ongoing aerial	9 hrs flying	26	0
Initial line sweeps	85 man-hours	126	5
Initial dispersed hunt	362 man-hours	382	25
2nd line sweep + dogs	97 man-hours	13	9
2nd dispersed hunt + dogs	424 man-hours	13	22
Pig snares/shooting	20 man-hours	NA	7
Judas goats	20 man-hours	0	NA
Follow-up survey #1	unknown	1	0
*Follow-up survey #2	210 man-hours	4	0
*Follow-up survey #3	380 man-hours	1	0
*Follow-up survey #4	380 man-hours	0	0

* Includes hours of all wildlife biologists surveying and shooters.

Aerial shooting

The helicopter (Macaw Helicopter Services, Saipan, R. Crowe-pilot, flying a Hughes 500) was used for about three hours/day over the first seven days of the operation beginning on 1 January 1998, and sporadically thereafter until 22 February 1998. In total 370 goats were shot from the helicopter (41% of the population) in a total of 30 flying-hours, of which 344 were killed during the first seven days in 21 flying-hours.

The overall plan was to shoot as many goats as possible from the helicopter and then follow this with ground hunting. No pigs were ever shot from the helicopter. One to two shooters and a spotter conducted helicopter operations. The spotter was useful in tallying numbers and keeping goats in sight when herds broke up. Shooters were armed with either a semi-automatic rifle with scope and a large-capacity magazine or a bolt action rifle with scope.

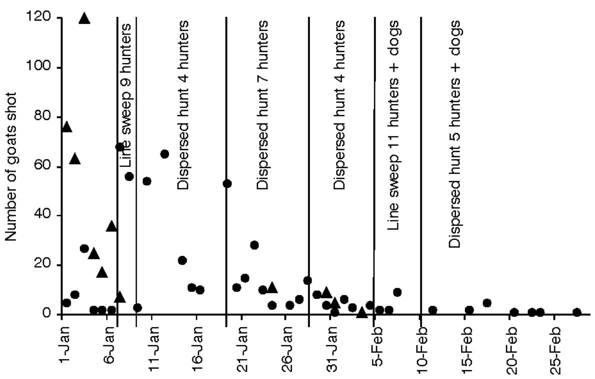

Fig. 1 Goats (*Capra hircus*) shot over time per operational tactic. Black triangles show the number shot from helicopters and black dots the number shot by ground hunting. Sarigan Island, CNMI.

Helicopter operations concentrated on those areas that had no or little canopy cover. The forested areas were searched, but proved too difficult to shoot through the canopy. Various methods of helicopter deployment were used. These included dropping off a shooter near a cave or dense vegetation and directing the shooter through helicopter surveillance.

Ground shooting

After the initial seven days of aerial hunting we began the ground hunting part of the operation targeting goats, pigs and feral cats. Our first tactic, used over the first two days, was to use a line formation of nine hunters, without dogs, that searched all forested areas.

Shooters were deployed in a skirmish line spaced roughly 30 yards apart moving across-slope following the general contour. All personnel were in constant communication through the use of hand-held radios. As shooters moved through the forest they stayed in visual contact with the person on either side, helped by the safety requirement to wear orange vests and hats. If they became separated due to terrain or dense vegetation or if they spotted animals, they would use the radios to communicate a desired coordinated action (i.e. stop or slow down). At the end of the day all shooters were debriefed to record the number of animals shot, number seen and in what area, tactical points, and any other items of interest. These line formations worked well for goats but not for pigs because the latter tend to break back through the skirmish line making a shot dangerous.

After two days of this large line-shooting operation, the crew was reduced to about five shooters (varied from one to seven). They were deployed in various ways to best exploit the terrain and goat behaviour. Often they would target a specific area, set up one or more spotters/shooters at a strategic point (i.e., a choke point or exit trail), then the rest would disperse through the area. This worked well for the small isolated groups of goats left. Again, radios were essential to keep information flowing as to the progress being made and if help was needed. Positions of the various shooters were continually monitored to ensure coverage and safety. This second tactic continued for about three weeks, after which the crew was again increased to 11 shooters and line sweeps were repeated. At this time we also used dogs. Trained to chase pigs, these dogs also helped with the goats. This greatly increased our coverage and allowed us some certainty that an area was goat or pig-free. After five days of clearance by the second round of line sweeps, the crew was again reduced to five persons. These five were dispersed singularly about the island with the primary mission of detecting goats or pigs. Once detected, help was then radioed for and a coordinated action deployed. The goats' behavioural trait of staying in a home range was exploited and made them vulnerable to this type of operation. Often goats could be detected by the bleating of the kids. Also the goats' habit of going into open areas to feed made them more observable. It should be cautioned that goats could detect loud radios and would flee. Our biggest problem in finding goats was their use of caves. We could sweep an area, but could never fully check all the caves and crevices in the volcanic terrain. Toward the end of the project we would smooth the ground along trails and in caves to be able to better detect animal tracks.

Goats were found in all habitats other than about 100 ha of swordgrass, giving an effective density of two goats/ha. This grass grows in dense clumps to about 3 m in height. As its name implies, it can inflict many slight cuts to exposed skin. This unpleasantness, plus its density and stifling heat, render it practically impenetrable and make for extremely disorienting and strenuous hiking conditions. Thus it was decided to burn off the *Miscanthus*. This was consistent with local agricultural practices. The two main reasons for doing so was to deny fleeing animals cover and allow easier traversal by foot (access into remote areas). *Miscanthus* was usually ignited with a flare shot from a marine signal gun out of the helicopter. In most areas the *Miscanthus* burned quickly, and did not spread into forested areas. It was our practice to go and investigate any carcass we found in the burned areas. We were curious to know if any animals died from the fire. We were always alerted to dead animals by the flies. Our only discoveries were of monitor lizards and geckos. As these were not totally consumed by the fire, it is assumed that anything larger would have left a greater proportion of carcass and would have been discovered more readily. It is therefore our conclusion that no large animals or birds perished in this manner.

Pigs

Pigs were restricted to the native and coconut forested areas, about 162.5 ha (Morton *et al*. 2000), giving an effective density of 0.42/ha in these habitats. For the first 30 days pigs were hunted opportunistically by personnel, including baiting and snares. This accounted for 54% (37) of the total pigs. At the end of 30 days we concluded that pigs were not being removed efficiently enough to bring about a total eradication. Dogs arrived on 3 February 1998 and began hunting immediately. Hunting with dogs accounted for the last 46% (31) of the total. The last pig was killed on 28 February 1998.

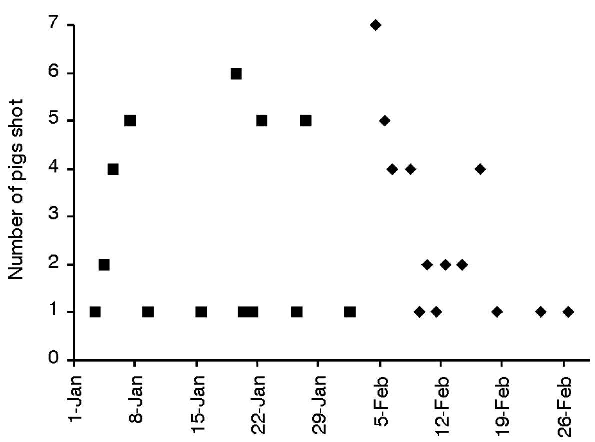

Fig. 2 Pigs (Sus scrofa) shot by hunting without dogs (black squares) and hunting with dogs (black diamonds) over project duration. Sarigan Island, CNMI.

Control techniques

Snaring for pigs

Snares were deployed but found to be of limited use. Two kinds were used, a commercial wire snare and a locally-constructed foot snare used with bait. The commercial wire snare was a constricting type equipped with a one-way continually tightening device. These are placed on trails and require that the animal put its head into it. These were not effective due to the lack of defined pig trails. The local snare was moderately successful. This snare was tied to an elastic tree branch, which would be bent down and fastened to a trigger. The snare loop would then be placed on the ground in front of a small semi-circular enclosure made of cut tree limbs that held the bait. The pig would stick its head into the enclosure's opening for the bait while standing in the snare's loop. The pigs feeding would release the trigger, the snare would tighten on one foot, and the branch's action would suspend the foot in the air thus holding the pig. Bait was split coconut. A more effective method was to bait a large area with split coconuts and return at night with the use of a spotlight to shoot any pigs feeding. For the first 30 days snaring, shooting over bait, and hunting were the only methods used for pigs. We had just about reached our limit; that is to say that we were no longer removing pigs, when the pig dogs were brought to the island.

Dogs

The pig dogs used were a local nondescript breed (mutt or "boonie" dog), about 34 kg apiece, and came from Alamagan Island in the Mariana chain. Dogs worked well, but could only chase a maximum of four pigs in a day before they were tired out. Two pigs in a day was more the norm. Chasing down pigs with dogs is a high-adrenalin pursuit that requires the shooter to strip down to the bare essentials. All gear must be tightly attached to the person, and protective eyewear is recommended. Gear consisted of a radio, canteen, and a short-barrelled semi-automatic rifle. As soon as the dogs signal that they have detected an animal, the shooters must start to run in the general direction so as to keep up. This physically exhausting and mentally nerve-racking run eventually ends in a limited access area. The pig's habit is to run until tired, then turn and face the dogs. When they do this, some natural barrier (such as thick brush or a ravine bank) usually protects their back. The shooter, upon arriving at the scene, is usually found to be blocking the exit path. As soon as the pig becomes aware of the person, it will try to run again and will charge through anything in its way. In this way the shooter is often attacked. Usually the shooter can dispatch the pig before it knows a person is there by slowly approaching. If not, then the dogs will charge in at the movement of the pig and attempt to hold it. This all makes for some seconds of utter confusion. After the pig is shot, a long break is required to rest the dogs and calm the shooter. The dogs usually held smaller pigs. These dogs would also detect and chase goats, although they did not seem as enthusiastic and would stop chasing once the goats reached an area of cliffs, boulders, or sharp rocks.

Radio-collared goats

On 12 February six radio-collared goats were released. These were airlifted to different sections of the island. They were heavily marked with water-soluble pink dye on their hair, carried pink flagging in their horns, and a pink-flagging collar for identification at long range. It was feared that these animals would be inadvertently shot, thus the identifying markers. These feral goats were obtained from Anatahan Island. They were deployed for about two weeks and finally removed at the end of the project. The decision to remove the radio-collared goats was made because it was unknown when or if a next trip to the island would be undertaken.

Rifles

Two general types of rifles were used. One was the Ruger semi-automatic model mini-14 ranch rifle in .223 calibre with five, 20, and 30 round magazines. The other type was the Ruger bolt action model M-77 in .220 swift calibre with a four shot magazine and fitted with a 3-9 variable Bushnell scope or a 6-20 variable Simmon scope and a bipod. CNMI law regulated the calibre of rifle used.

Communications

Hand-held radios were a critical link throughout the operation, both for person-to-person and person-to-helicopter communications. Hand-held radios provided coordination between personnel and provided the flexibility to adapt to any situation. It also increased our safety factor by both preventing dangerous shooting conditions and facilitating rescue for possible accidents. Each person carried a unit plus a spare battery. Some of the units failed due to the high humidity and by getting soaked with perspiration. Waterproof or "integrally safe" type radios are double the standard price, but they may be well worth obtaining. We did lose or otherwise destroy a few units and it is recommended to have extras on hand. Power from a gas generator was used for re-charging. For field operations, radios should be firmly attached to the shooter in a manner that is easy to use (i.e. close to the face) but secure. Plastic ziplock freezer bags or waterproof radio bags should be carried to cover units during rain.

Communications between Sarigan and Saipan (location of closest facilities) were indispensable. A single side band radio was used to communicate with a 24-hour operator (Emergency Management Office) on Saipan. We could also talk directly with Macaw Helicopter's home base. A long-range radio proved to be a beneficial requirement for logistical planning, weather updates, and emergencies.

Boat directed

A small (4.6m) boat equipped with a 12hp outboard was used to move materials from supply boats to shore. It also served as a spotting platform to direct shooters onto animals and to land shooters at otherwise inaccessible areas around the island. Typically, the small boat would depart with an operator and a spotter. Binoculars and radios were needed. The boat would slowly move around the island searching for animals on the steep slopes. Once found, locations would be radioed to ground shooters who had been previously deployed about the island. During these outings we would occasionally see goats close to shore in otherwise inaccessible areas. The small boat would nose in and drop a shooter off. Again radio communications were of the utmost importance. Shooting from the boat was also tried, but the ocean swell made this impractical and usually ended in expended ammo with no gain.

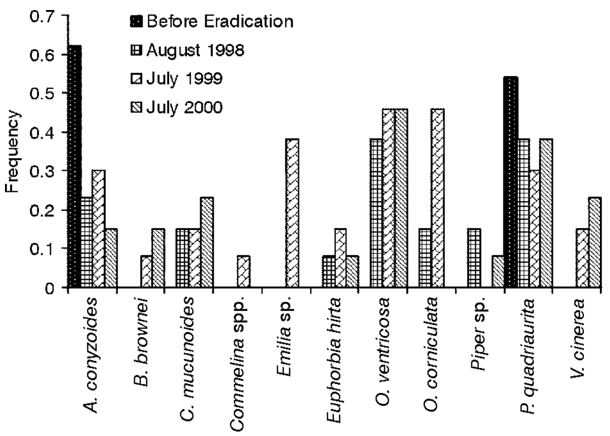

Fig. 3 Frequency of herbs and weeds on 13 plots before and after feral ungulate eradication. Sarigan Island, CNMI.

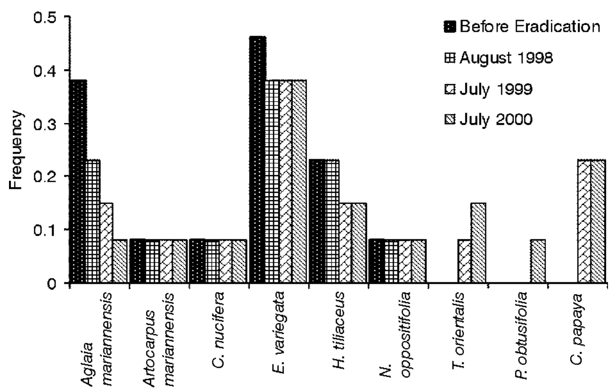

Fig. 4 Frequency of tree species found on 13 plots before and after feral ungulate eradication. Sarigan Island, CNMI.

Changes in wildlife and vegetation

Vegetation

The vegetation responded immediately to the removal of ungulates. The total number of plant species found in all 13 plots has increased from seven in 1997 to 17 in 1998, 22 in 1999, and 25 in 2000.

Many of these new species are herb and weed species (one species in 1997 vs 11 in 2000). Two species originally found on the plot, the fern *Pteris quadriaurita* and the colonising weed *Ageratum conyzoides* have both declined (Fig. 3). Species found on the plots since 1998 include; *Blechum browneii*, *Calopogonium mucunoides*, *Operculina venricosa*, *Euphorbia hirta*, *Oxalis corniculata*, *Oplismenus undulatifolius*, *Veronica cinera*, *Emilia* sp., *Piper* sp., and *Commelina* sp. These species are widespread throughout the tropics and have been observed in the Mariana Islands since the early 1900s (Merril 1981).

Grasses and sedges have increased from one species in 1997 to four in 2000. Tree species have increased from a total of four in 1997 to nine in 2000 (Fig. 4) and have shown a steady increase in the number of seedlings on the plots. These tree species are *Aglaia mariannensis*, *Artocarpus mariannensis*, *Cocos nucifera*, *Erythrina variegata*, *Hibiscus tiliaceus*, *Neisosperma oppositifolia*, *Trema orentalis*, *Premna obtusifolia,* and *Carica papaya.* All are native species to the Marianas, except for *C. papaya*, which is from the Americas but considered naturalised (Raulerson and Rinehart 1991).

Along with the increase in vegetation there has been an increase in canopy cover and a decrease in ground cover (from shading) on those plots near or within the forest. Plots located on grasslands have shown an increase in ground cover from near zero (bare dirt) in 1997 to 100% cover in 2000.

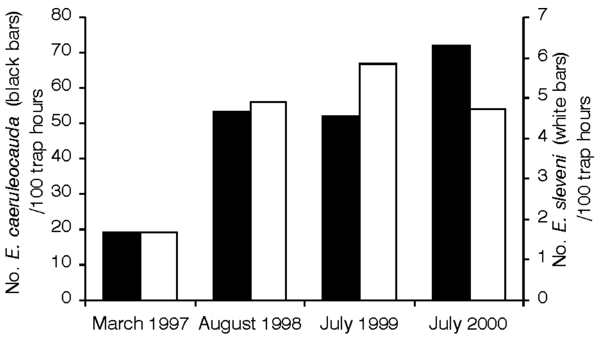

Fig. 5 Catch rates (# lizards/100 trap hours) for *Emoia caeruleocauda* and *Emoia slevini* before and after feral ungulate eradication. Sarigan Island, CNMI.

Wildlife

Fruit bats' populations appear to be stable at about 175 and showed no apparent changes. It is assumed that fruit bats are linked to the number of flowering/fruit producing trees, so that any change in bat numbers would require a number of years until the trees mature. Likewise there is no apparent change in rat populations, although only a few rats (<6/year) were ever caught. Rat trapping was done in 1999 and 2000.

There were no significant changes in bird detections. There appears to be a slight increase in the number of megapodes detected from about one per station to two per station. Kingfisher (*Halcyon chloris*) populations also show a slight increase. Micronesian starling (*Aplonis opaca*) detections went up in 1999 but decreased in 2000. Honeyeater (*Myzomela rubratra*) detections have declined in 2000. It is likely that it is too soon to see changes in these species and the slight variations detected could be due to seasonal foraging differences, observer variance, or to El Niño/La Niña events.

Catch rates for blue-tailed skinks (*Emoia caeruleocauda)* and the endemic Slevin's skink (*Emoia slevini)* have greatly increased (Fig. 5).

DISCUSSION

Eradication

Goats

The helicopter shooting worked well as a means to remove large numbers of goats in open or steep terrain. Calculations show that we killed 38% in approximately 21 hours of shooting. After this the numbers shot dropped off sharply. The goats quickly learned to recognise the sound of the helicopter. The helicopter was also useful in surveying remote areas for the absence of goats.

Line sweeps with a number of shooters was a good way to clear dense sections of forest. At first try, the line was difficult to keep straight and gaps and holes would form presenting dangerous conditions and incomplete coverage. Once the method was practiced a number of times it became very efficient. Walking in a line would force the animals to either try to get through or go around the top or bottom of the line. Pigs usually went through, goats went around. Spotting goats using binoculars and directing shooters to the location by radio was another method that was successful; especially in very steep terrain when the spotter could be offshore on a boat.

The "Judas goats" (Taylor and Katahira 1988) did not work for us. In reviewing the project, I believe that the failure of this operation was due to association problems between goats already present (with established herds and ranges) and the newcomers. The collared Anatahan goats were bigger and hairier then the small short-haired goats on Sarigan. I suspect that over a longer period of time they would have joined up, but in our short period it didn't happen. It has been proposed that the dye and flagging alone could have been the deterring factor, but this technique (using flagging) has been used since with satisfactory results (pers. obs.). For that project we used goats from the same island (Norman Island-British Virgin Islands). It would have been better for us to collar Sarigan Island goats (in their local ranges) immediately upon commencement of operations. It is probable that some animals would have been shot inadvertently, but with high visibility markings this could have been kept to a minimum. Also, if goats had been released early in the project, it is felt that they would have joined up with survivors quicker, thus becoming more in tune to the best hiding spots.

Pigs

The local type of snare worked well in a limited capacity. It was a low-cost method of capturing pigs. Shooting over bait at night worked very well when pig concentrations were high and they were still naïve about humans. The pig dogs, however, were the most efficient means of finding and killing pigs. At the time the dogs came in, we were not killing additional pigs. They had learned to escape us and it would have been difficult to kill the remaining 31 without the dogs.

Logistics

Our most significant problem was the movement of consumable supplies such as water, gas, and food to the island. This used up much of our budget in the form of helicopter time. Local fishing boats were cost-efficient, but hard to schedule and were weather-dependent. Food was not as big a problem as drinking water and gasoline. We did have rainwater catchments established, but it rained little during the project's duration.

Supply procurement and coordination by the project supervisor was an added burden that detracted from smooth running field operations. It is recommended in the future that a supply officer be set-up that is intimately familiar with the needs of the island crew. This person's sole responsibility would be to plan, purchase and ship supplies to the island.

Equipment

All equipment should be first rate with plenty of spares on hand. Much gear will be damaged and should be replaced immediately to keep the shooters motivated and avoid dangerous situations; plan supply and shipping alternatives well before-hand and have alternate sources on standby. One item that I found to be of importance was military MRE's or Meals Ready to Eat. One of these each day for lunch allowed shooters to stay in the field and considerably boosted morale.

Rifles

Except for the close-up work of hunting with pig dogs, or clearing caves, my choice would be to outfit every rifle with a telescopic sight of at least 3x magnification. This scope could be used in most situations and would greatly improve the accuracy of the shooter. In thick brush the scope did not appear to be a hindrance. The problem faced was keeping them in working order due to the hard spills and humidity. By the end of the project we had used up all of our replacements (eight total). Spending money initially on expensive but rugged military specification (Mil. Spec.) scopes could mitigate this. To avoid being stuck in a situation where rifles can't be used due to lack of a working scope, some rifles with iron sights should be kept on hand.

Personnel

All field personnel must be physically fit and mentally prepared for hardships. Shooters should be experts in firearm use and hunting techniques. All should be motivated individuals picked for their ability to move over any terrain under harsh conditions carrying the necessary field equipment. A day walking through the forest is not only tiring but mentally nerve-racking due to the extreme terrain and constant life-threatening hazards from tripping and falling. All personnel should be familiar with the project goals and be committed to achieving them. Physical and mental stresses were a constant factor in shooter fitness. Every effort should be made to provide comforts in camp to allow shooters to relax. This could include: good chairs/recliners to rest sore muscles; a full time cook; and cold drinks. These amenities go a long way to keeping an isolated crew, working under harsh conditions, focused on the project.

Vegetation and wildlife

Vegetation

In the Marianas, it seems that goats and pigs are a destructive force capable of changing the microclimate of the forest floor. The destruction of the ground cover, the drying out of the forest floor, and the subsequent loss of soil through erosion kills the trees. The eating of new growth by ungulates suppresses any regeneration capabilities of the forest. The result is a steady decline of forest habitat whether it is native or coconut forest. This in turn has serious implications for native wildlife that require forest.

Removal of ungulates has stopped and reversed the process of forest loss and has re-established the forest floor microclimate (in less than a year). Likewise, the native forest is expanding and regenerating. Many new trees were observed growing in areas that were previously grassland; the sudden eruption of new growth has undoubtedly slowed erosion and contributes to new soil being laid down.

The general trend appears to be the changing of grassland to forest. Areas of *Chrysopogon acicularis* grassland and bare dirt are sprouting new thickets of the shrubby species *Dodonaea viscosa*. In some areas of the *Miscanthus* grassland new stands of *Morinda citrifolia*, *Tremna orientalis,* and *Leucaena leucocephala* (introduced species) trees have become established. Some of this growth in the *Miscanthus* could be a result of the combination of burning and ungulate removal.

The unknown side to this increase in vegetation is the explosion of the invasive vine *Operculina ventricosa*. This vine, which I had not observed prior to ungulate removal, now covers most of the native forest and surrounding *Chrysopogon acicularis* grassland in the central valley. It had been previously recorded as present on Sarigan (Fosberg 1979), but is a species that is readily consumed by goats. It is unknown what effect this vine is having on the regeneration of the native forest. The vine does not appear to do well in the coconut forest or *Miscanthus* grassland. Although it blankets the central valley, new tree saplings are still pushing up through the vine. On a more positive thought, it has acted as a band-aid on the heavily disturbed areas, shading the ground and trapping moisture. It seems to provide an increase in foraging habitat for lizards, and megapodes have been observed to use it as well. Only time and continued monitoring will reveal its impact. An observation of note on the vine, was clusters of leaf-footed or squash bugs (Family *Coreidae*) apparently feeding on it. Whether this species will act as a form of biological control remains to be seen.

Wildlife

The most rapid change of any wildlife species is the increase of skinks, mostly due to their high reproductive rate. The combination of an enlargement in forage area (increased vegetation) and the removal of a direct predator (the pig) has seemingly benefited these lizards.

The skinks themselves form a prey base, especially for kingfishers and megapodes. Indeed there seems to be an upward trend in these two bird species numbers, however it is probably too soon to tell. Unfortunately, these skinks probably function as a prey item for monitor lizards (*Varanus indicus* - believed by some to be introduced) and feral cats, which also prey on the native birds. It is interesting to speculate what affect an increase in these two species' populations will have on Sarigan.

Noticeable increases in bird populations, especially those that depend on mature tree species, will require a longer monitoring period. But it is probable that the increases seen in major food species such as the coral tree (*Erythrina variegata*) and papaya (*Carica papaya*) will be beneficial. Papaya had not been evident in the forest prior to eradication and now is abundant (a primary starling and fruit bat food). Coral trees, which produce a beautiful red flower, are now observed as a common sapling established in the *Chrysopogon* grasslands. This is the primary source of food for honeyeaters and fruit bats during the dry seasons (December-April). Bat numbers will require further monitoring to see if populations change. Although no definite increase can be stated now, fruit bats have been observed roosting in young *Tremna* trees that had previously been *Miscanthus* grassland. This expansion in roosting and foraging habitat can only be viewed as beneficial.

Rat populations have shown no increase although the data is cursory. On the neighbouring island of Guguan (no feral animals), the Pacific rat is extremely common and can be seen throughout the day (pers. obs.). On Sarigan (since 1996), rats were uncommon (rarely seen only at night); only a few were caught during surveys although this could be a function of trap placement and/or bait. It is speculated that feral cats and monitor lizards play a role in the reduced rat numbers on Sarigan. It remains to be seen what an increase in fruit will have on the rat population and consequently the cat and bird populations.

One species that must be mentioned is the dog dung fly *Musca sorbens.* Unfortunately no scientific insect data of any kind was collected and the evidence on this species is anecdotal. This species is considered a pest and had previously occurred in high numbers on Sarigan (pers. obs.). Food had to be covered. On the last follow-up trip almost no flies were encountered (<5) and food could be left uncovered. This species requires animal dung for its life cycle, and the apparent reduction in numbers is probably linked to the elimination of ungulates. Removing this species has important social-political implications for future projects in the Marianas and could be touted as a major benefit. Also, this species was observed to be a prey item for skinks. With an apparent decrease in flies and an increase in skinks, one wonders what might be filling the skinks' needs. Insect data needs to be collected for future projects.

Although the removal of feral goats and pigs has had some unforeseen consequences, the majority of the changes can only be viewed as positive. The project goals of stopping forest loss and increasing megapode (and fruit bat) habitat appears to have been achieved. This island serves as an important ongoing experiment in island wildlife management.

Future needs

Probably every eradication project could use more funding and professional help in documenting the changes. Removal of herbivores affects all aspects of an island's ecosystem and creates a large-scale laboratory for studying these changes. More complete surveys of the plant and wildlife communities of Sarigan and of neighbouring islands need to be undertaken to provide basic knowledge of the native species and their life histories. These surveys should be performed a minimum of twice a year to represent both the dry and wet seasons. Surveys must be standardised and be of a simple design so as to be readily reproducible. Data acquired needs to be organised and published for future biologists.

One of the missed opportunities of this project was to study changes in the nearshore marine community and its relation to feral ungulates and forest destruction. If another such programme is undertaken in the Marianas, it is urged that some marine transects be established to explore these relationships.

It is my opinion that the two most important aspects to achieving a complete eradication on Sarigan was momentum and communications. Momentum must be maintained once a project is started or the result will only be wasted time and effort. Few enjoy slaughtering the animals, and to do so to only have a project halted midway is lamentable. Two major considerations in sustaining momentum are politics and supplies. Without good support for either the project is questionable. The other major factor for us was communications in the field. Without effective and efficient ways to coordinate shooters we could not have achieved what we did in so short a period.

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The response of herbaceous vegetation and endemic plant species to the removal of feral sheep from Santa Cruz Island, California

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Abstract From 1984 to 1998 we monitored the response of herbaceous vegetation and endemic plant species to the eradication of feral sheep from the western 90% of Santa Cruz Island, California. Total herbaceous cover increased and bare ground decreased after sheep were eradicated from the island. Alpha diversity of herbaceous vegetation reached a maximum in the two years prior to the end of eradication and the first two years after eradication, then declined. The number and relative frequency of native herbaceous species were inversely related to increased herbaceous cover and the relative frequency of alien species. The reduction in number of native species was due to alien species that already occurred in an area rather than with alien species invading the area. Thirty-three of the 43 endemic plant species on the island showed an increase in distribution and/or abundance following the eradication. New populations of two of the five rarest species on the island were discovered within seven years of the end of the eradication programme, and abundance of these two species increased. Of the other three species, the distribution and abundance of one remained unchanged while the two other species showed alarming declines. The declines of these two species were attributed to a proliferation of alien grasses and impacts from feral pigs. Because grasslands occupy almost 50% of the area of the island, the response of the herbaceous vegetation was relatively undesirable from a conservation perspective. But most of the endemic species showed positive responses, and other studies on Santa Cruz Island indicate that communities on the island that are dominated by shrubs and trees appear to be showing rapid rates of recovery from sheep impacts. A variety of outcomes can be expected to occur as a result of eradication of large numbers of grazing ungulates from islands, so eradication programmes should only be considered the first step in a long process of restoration rather than an end in themselves.

Keywords alien plants; eradication; feral animals; grasslands; islands; monitoring; restoration.

INTRODUCTION

Invasions by alien species have been one of the most important forces altering biotic and abiotic processes on islands (Loope and Mueller-Dombois 1989). Predation by invasive alien animals has changed trophic structure and been a direct cause of extinction, while intense herbivory has drastically altered structure, species composition, and function of ecosystems, as well as been a major contributing factor in extinction (Savidge 1987; MacDonald *et al.* 1989). Alien plants have also modified species composition and ecosystem function (Mack and D'Antonio 1998), and although there are no documented cases of native species being driven to extinction as a direct result of interactions with alien plants, there is concern that this could eventually occur (Cronk and Fuller 1995).

Preserving or restoring native species and natural communities is one of the primary goals of virtually all invasive species control and eradication programmes. The goals are usually stated in terms of conserving or increasing diversity, although diversity is seldom explicitly defined. In situations where the decimation of an entire group of native organisms by an introduced predator is occurring (e.g. the effect of the brown treesnake *Boiga irregularis* on the avifauna in Guam; Savidge 1987) the definition of diversity becomes obvious; reduction in gamma diversity is assured with extinction. However, in situations where extinction has not occurred but the relative abundance of native species is declining, diversity becomes a more complex issue. In this case eradication or control of an alien species will not affect gamma diversity, but both alpha and beta diversity can change. The desired outcome is that diversity will increase, but since most areas are invaded not just by one but by a suite of aliens (especially islands; Loope and Mueller-Dombois 1989), the question of what species are responsible for any changes in diversity becomes important.

Alien plants and animals have heavily invaded the California Channel Islands (Coblentz 1980; Van Vuren 1981; Junak *et al.* 1995). The greatest impacts have been by feral animals, primarily sheep (*Ovis aries*), goats (*Capra hircus*), and pigs (*Sus scrofa*) (Coblentz 1977, 1978, 1980; Van Vuren 1981, 1984). These impacts were especially severe on Santa Cruz Island (SCI), where sheep had been introduced in the early to mid 19th century (Van Vuren 1981). The sheep were mostly feral by the early 1870s, and more than 50,000 sheep were estimated to be on the island in the 1890s. Attempts were made in the 1900s to

Pages 141-154 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

control the sheep population by trapping and shooting, but the efforts were not successful (Van Vuren 1981). By the 1980s, there were an estimated 20,000 sheep on SCI (Van Vuren 1981). The density was more than double that of the maximum stocking rates of mainland sheep operations, and over one-third of the island was classified as being heavily impacted (Van Vuren 1981). This resulted in an increase in bare ground and subsequently higher erosion rates, decreased herbaceous vegetation, reduction and modification of shrub communities, and a decrease in abundance and diversity of birds (Brumbaugh 1980; Hobbs 1980; Hochberg *et al.* 1980; Minnich 1980; Van Vuren 1981).

Beginning in late 1981, The Nature Conservancy (TNC) undertook a programme to eradicate feral sheep from the western 90% of SCI that it managed. The goals of the programme were to preserve, protect, and enhance the natural systems, flora, and fauna of the island (Schuyler 1993). The eradication was completed successfully in 1988 after 36,000 sheep were shot off the western part of the island (Schuyler 1993). An estimated 1000 – 5000 sheep remained on the eastern 10% of the island (Klinger unpubl. data), so a control programme using a combination of fencing, drives, and shooting was implemented in 1989 to prevent incursions onto TNC land. An additional 4100 sheep were shot during the control programme, which continued into early 1997 (Klinger unpubl. data). During this time one band of three sheep was found 10 km from the boundary fence, but virtually all of the rest (>97%) were shot within 1 km of the fence. A small number of other groups were within 7 km of the fence, but were never resident for more than one month. The National Park Service acquired full ownership of the eastern 10% of the island in 1997, and in the next three years removed 9000 sheep from the eastern part of the island. There is cautious optimism that feral sheep no longer occur on SCI (K. Faulkner, Channel Islands National Park, pers. comm.).

In addition to the eradication and control programmes, monitoring programmes were established to evaluate the response of grassland vegetation and endemic species to the sheep eradication. A preliminary analysis (Klinger *et al.* 1994) indicated that there were changes in species composition in grasslands following eradication of the sheep, but these changes were neither systematic nor entirely predictable. In addition, removal of cattle and feral sheep was a likely trigger for the rapid expansion of fennel (*Foeniculum vulgare*), a highly-invasive herbaceous species from the Mediterranean (Brenton and Klinger 1994). In this paper we extend the prior analysis (Klinger *et al.* 1994) to examine in greater detail factors influencing changes in diversity and species composition in the grassland community, and also present data for the response of endemic species to the eradication.

STUDY AREA

Santa Cruz Island is the largest of California's eight Channel Islands (Fig. 1). It lies approximately 40 km from the coast, and with a land area of over 250 km^2 it is considered the most topographically and ecologically diverse of the California islands (Junak *et al.* 1995). Like the rest of the California islands, evidence indicates that SCI has never been connected to the mainland (Vedder and Howell 1980). Consequently, it is considered a fringing rather than a continental island (Moody 2000).

Santa Cruz Island is divided along its long axis by a central valley flanked by two east-west tending mountain ranges. Six major vegetation communities occur on SCI, including grasslands, chaparral, woodland, coastal scrub, pine forest, and riparian (Philbrick and Haller 1977; Minnich 1980). A total of 650 plant taxa occur on the island, of which 26% (n=170) are alien (Junak *et al.* 1995). Despite their proximity to the mainland, the Channel Islands are known for having relatively high levels of endemism (Raven 1967). There are 43 species of island endemics on SCI, of which eight occur only on the island (Junak *et al.* 1995). Two taxa are considered to have gone extinct in the last 100 years (Junak *et al.* 1995), and although feral animals are considered to have played a role in these extinctions the degree to which they were involved is unknown.

Besides feral sheep, cattle and feral pigs have been resident on the island for >150 years. Except for a remnant herd that has been allowed to remain for historical rea-

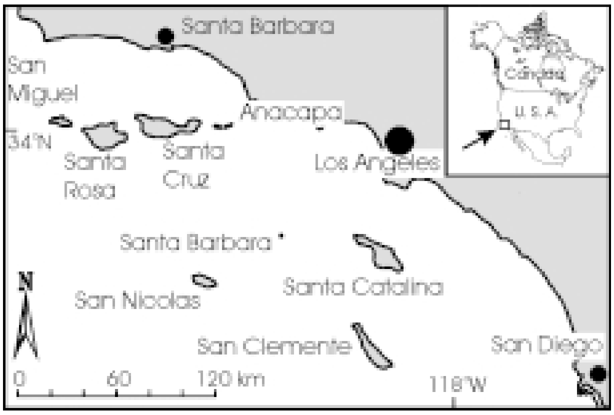


Fig. 1 Santa Cruz Island and the California Islands.

Table 1 Annual rainfall recorded on Santa Cruz Island (Central Valley), 1984-1993.

Year	Rainfall (cm)
1984	25.4
1985	25.0
1986	49.5
1987	21.8
1988	24.3
1989	13.9
1990	10.0
1991	24.3
1992	32.9
1993	39.5

sons, the cattle were removed in 1988. Pigs occur across the island, and estimated numbers fluctuated from 800 to 4500 between 1991 and 1998 (Klinger unpubl. data).

The Mediterranean climate is modified by the surrounding maritime conditions. Winters are cool and wet, late summer, spring and fall are clear and warm, and early summer is foggy and cool. The 90-year average rainfall is 30.7 cm (L. Laughrin, U.C.S.B. Natural Reserve System, unpubl. data), with about 90% of the precipitation occurring from November-April. Rainfall can vary in different parts of the island, but these patterns tend to be constant from year to year. With the exception of 1986, a series of dry years occurred from 1984-1991 (Table 1).

Chumash Indians occupied SCI from approximately 7000 YBP to the early 1800s. Ranching and agriculture were the primary land use from the mid-1830s to 1977. TNC acquired an interest in the island in 1978 and full ownership of the western 90% in 1988. TNC managed the island as an ecological reserve through 1998, when it began to transfer most of the ecological management to Channel Islands National Park (CINP).

METHODS

Grassland Community

Data were collected from 21 plots in grassland habitat sampled in seven springs (March-May) between 1984 and 1993 (Table 2). Three to five 30 m transects were randomly selected along a 30 m baseline within each plot. Twenty 0.25 m^2 quadrats were spaced equidistantly along each transect, and the presence or absence of all herbaceous species was recorded within each quadrat (U.S. Forest Service 1983). Ocular estimates of the total percentage cover of live vegetation, litter (dead or dry organic material) and bare ground were made for each quadrat.

The estimated density of sheep were used to classify the plots as being in areas with high, moderate, or light impacts (Appendix 1). The density was estimated from the total number of sheep killed in an area (Schuyler 1993).

Table 2 Sampling effort for monitoring the response of grassland vegetation to the removal of feral sheep on Santa Cruz Island, California. Sampling was not conducted in 1986, 1988, and 1990. There were twenty 0.25m² quadrats/transect.

Year	Number of Plots	Number of Transects
1984	21	87
1985	21	87
1987	8	31
1989	10	42
1991	11	24
1992	18	54
1993	19	48

Vegetation species were initially grouped into seven different guilds: native annual grass, native annual forbs, native perennial grass, native perennial forbs, alien annual grass, alien annual forbs, and alien perennial forbs. Because alien perennial forbs and native annual grasses occurred infrequently, analyses were done for four guilds: alien grass, alien forbs (annual and perennial), native grass (annual and perennial), and native forbs.

Nested frequency data were analysed by summing the total number of occurrences across all four nests for each species for each year in each impact class. In 1987 only one plot was sampled in the medium-impact class and none in the low-impact class, so 1987 was omitted from the analysis. A disproportionately small number of low and medium-impact class plots were sampled in 1989 and 1991, so these classes were combined for analysis in all years. Differences in frequency of occurrence between guilds, impact, and years were first analysed with a log-linear test. Because the fully-saturated log-linear model was significant, Chi-square analyses were used to test differences in frequency of occurrence in different subsets of the full model. Because of the non-independence of multiple comparisons, a Bonferroni adjustment was used to guard against inflated type 1 error and determine significance of each of the comparisons (Sokal and Rohlf 1981).

The proportion of quadrats within each plot that a species occurred in was used as a measure of abundance for that species within the plot. Frequency values were summed for each species and standardised as a proportion of the total number of quadrats the species could have occurred in (Smith *et al.* 1987). Because a single number does not always adequately describe diversity patterns (Magurran 1988), we derived different indices of alpha diversity for each plot: species richness (N_0), diversity (N_1, and N_2; Hill 1973), and Molinari's index of evenness (Molinari 1989). N_1 was calculated as exp H' where H' is Shannon's diversity index and N_2 as 1/C where C=Simpson's index of concentration (Hill 1973). Because of a strong correlation between N_1 and N_2 ($r = 0.98$) only N_0 and N_2 were used for analysis.

Diversity indices are known to be sensitive to sample size (Magurran 1988), and unequal sampling of the plots occurred during the study (Table 2). To test if the unequal sampling effort was leading to spurious diversity patterns, 30 runs were made for randomly-selected values of S, N_2, and E for sample sizes ranging from 1–18 in the years 1984, 1985, 1992, and 1993. Regression analysis was then used to test if there was a correlation between the number of plots sampled and the indices of diversity. A separate analysis was done for each of the four years. There was no significant correlation between the diversity indices and the number of plots sampled for any of the years, suggesting that differences in diversity were not an artefact of sampling intensity.

Similarity in total species composition between years was analysed with the Morisita-Horn index (Magurran 1988). Cluster analysis (Gauch 1982) was used to construct dendrograms to show relationships in species composition between years. Distances were based on chord Euclidean distance, and average linking was used as the linkage method (Sneath and Sokal 1973). Kendall's coefficient of concordance (Conover 1980) was used to test the rank abundance between years of the ten species with the greatest proportional occurrence values. Whittaker's index (Pielou 1975) was calculated for each plot and used to evaluate the turnover of species between years.

The relationship between the indices of alpha diversity and site characteristics was analysed with multiple regression procedures. Independent variables that were used in the analyses included: (1) the initial density of sheep in an area prior to the eradication, (2) the estimated density of sheep in the different years of vegetation sampling, (3) the number of years prior to and after eradication, (4) four different topographic measures (slope, aspect, elevation and cover of bare ground), and (5) two biotic variables (vegetation cover and percentage cover of organic litter). The relationship between the site characteristics, species richness, and relative frequency of the different guilds, was also analysed with multiple regression procedures. Least-squares regression was used to analyse the univariate relationship between alpha diversity and rainfall. Because least-squares and multiple regression analyses are sensitive to outliers, cases with Cooks distance >2.00 were removed from the analyses. Residual plots were inspected for normality, equality of variance, independence, and linearity.

Canonical Correspondence Analysis (CCA) was used to analyse the relationship between species abundance and site characteristics. The site characteristics were the same as described above, and the year of sampling was used as a covariate (ter Braak 1995). Forward-stepping multiple regression was used to select the variables that had a significant correlation with species abundance patterns. Rare species were downweighted in the analysis.

Endemic Species

Systematic monitoring of endemic species began in 1991. Three different approaches were used: general searches, estimates of cover and/or density in plots, and focused monitoring of five species highly restricted in distribution and/or abundance. General searches for all endemic species were conducted in 1991, 1992, 1995, and 1996. The number of observations of each species in each of 190 x 1km^2 blocks was noted each year. Because only half the blocks were surveyed each year, the searches were divided into two time periods: 1991–1992, and 1995–1996. Changes in the frequency of the total number of occurrences between time periods of all endemic species were analysed with Chi-square tests.

The density and/or cover of the more-widely distributed and abundant endemic species were estimated from 1991 to 1995. One hundred plots were located from near the eastern end of TNC's property to the west end of the island. The plots were 30 m x 2 m in dimension, and covered all of the island's major geographic subdivisions except the north-west side. Sixty-one plots were initially established in the spring of 1991, but following a preliminary sample size analysis another 39 were added during the spring of 1992. Plots were stratified proportionally by the area of the five major plant communities (Minnich 1980).

Most plot locations were picked by randomly selecting a starting point on the eastern end of TNC property, then stopping every 0.8 km along a road and picking a random compass bearing and distance within 150 m from the road. This system allowed for randomness but also let field staff have a consistent reference distance to begin searching for plots. Seven plots were located along trails; a random compass bearing and distance were used to select these locations, but distance between location markers varied.

All species (endemic and non-endemic) occurring in a 2 m wide belt (1 m on either side of the tape) along the tape were recorded. The number of shrubs and trees rooted within the 2 m wide belt were counted, and cover was estimated with the point-intercept method (Bonham 1989). Cover estimates were made by vertically extending a thin metal rod at 100 points spaced equidistantly along the tape. All species of plant the rod intercepted were recorded, and the height of the tallest species intercepted by the point was also recorded. Sampling occurred from March to May each year from 1991 to 1995.

Five endemic plant species with highly restricted distribution and/or abundance were surveyed annually from 1991 to 1998. *Arabis hoffmannii* Rollins (Brassicaceae) is a herbaceous perennial with only three known populations on SCI. One population of *A. hoffmannii* is also known to occur on Santa Rosa Island. *Berberis pinnata* Lagasca subsp. *insularis* Munz (Berberecidae) is a woody perennial species that occurs on SCI and Anacapa Island. It is known to occur in three areas of SCI. *Dudleya nesiotica* A. Berger (Crassulaceae), *Malacothamnus fasciculatus* Greene subsp. *nesitoicus* Kearney (Malvaceae), and *Thysanocarpus conchuliferus* Greene (Brassicaceae) occur only on SCI. *M. fasciculatus* subsp. *nesioticus* is a woody perennial that had only one known population on the island. *T. conchuliferus* is a herbaceous annual with 14 historic locations on the island. *D. nesiotica* is a perennial succulent restricted to a 100 ha area on the western tip of SCI. Historically it had been reported to be relatively abundant within its range (Hochberg *et al.* 1980).

All individuals of both *A. hoffmannii* and *M. fasciculatus* subsp. *nesioticus* were marked, and measurements of height and counts of the number of stems, flowers and/or fruits and were made annually. Searches were made of known locations of *B. pinnata* and *T. conchuliferus*, and presence/absence noted in each location. Counts and estimates of cover of *D. nesiotica* were made in 30 x 0.25 m^2 quadrats from 1991 to 1995, and 70 quadrats from 1995 – 1998. Least-squares regression was used to analyse changes in stem number and/or flowers and fruits for *D. nesiotica*, *A. hoffmannii* and *M. fasciculatus* subsp. *nesioticus*.

Analyses of contingency tables and regression analyses were done with Systat 8. Canoco 4 was used to conduct the CCA (ter Braak and Smilauer 1998). All statistical tests were considered significant if $p < 0.05$. If $0.10 > p > 0.05$, the test was considered to be marginally significant.

Nomenclature follows Junak *et al*. 1995.

RESULTS

Grassland community

A total of 161 species occurred in the plots (Table 3). Based on the percent of all species recorded in the plots, native species occurred disproportionately less than would have been expected ($p < 0.0001$). Although they made up < 32% of the species, alien forbs and grass accounted for >68% of the frequency of occurrence of all species in the quadrats ($p < 0.0001$).

Table 3 Relative frequency of occurrence of five different guilds of plants in relation to the number and percentage of species within each guild on Santa Cruz Island, California, 1984-93.

Guild	Species (N)	Species (%)	Frequency (%)
Native Forb	85	52.7	26.5
Native Grass	10	6.2	3.9
Shrubs	15	9.3	1.5
Alien Forb	34	21.1	28.5
Alien Grass	17	10.7	39.6

Table 4 The frequency of occurrence (%) for species in four guilds of herbaceous vegetation in areas with different impacts from feral sheep on Santa Cruz Island, California. Frequency of occurrence was summed across nest sizes of 25 cm², 625 cm², 1250 cm² and 2500 cm² within 2500 cm² quadrats for each species in each guild. Values are the percentage of occurrences for each combination of guild, impact, and year divided by the total for the entire study.

Year	Alien Forb	Alien Grass	Native Forb	Native Grass	Total
High Impact					
1984	4.2	17.6	1.5	0.1	23.5
1985	7.0	12.9	4.5	0.1	24.5
1989	2.6	7.8	1.0	0.0	11.3
1991	1.3	4.4	0.3	0.0	6.0
1992	3.7	18.9	3.9	0.5	27.0
1993	0.6	6.9	0.2	0.1	7.7
Total	19.3	68.4	11.4	0.9	100.0
Medium/Low Impact					
1984	1.4	20.7	0.2	0.5	22.7
1985	5.1	16.6	1.8	0.6	24.0
1989	1.4	7.5	0.2	0.1	9.2
1991	5.8	7.2	1.3	0.0	14.2
1992	6.5	13.4	2.0	0.5	22.3
1993	0.4	6.7	0.3	0.1	7.5
Total	20.4	72.0	5.7	1.8	100.0

Alien annual grass had significantly greater frequency of occurrence in all combinations of impact category and years than other herbaceous vegetation guilds (Table 4).

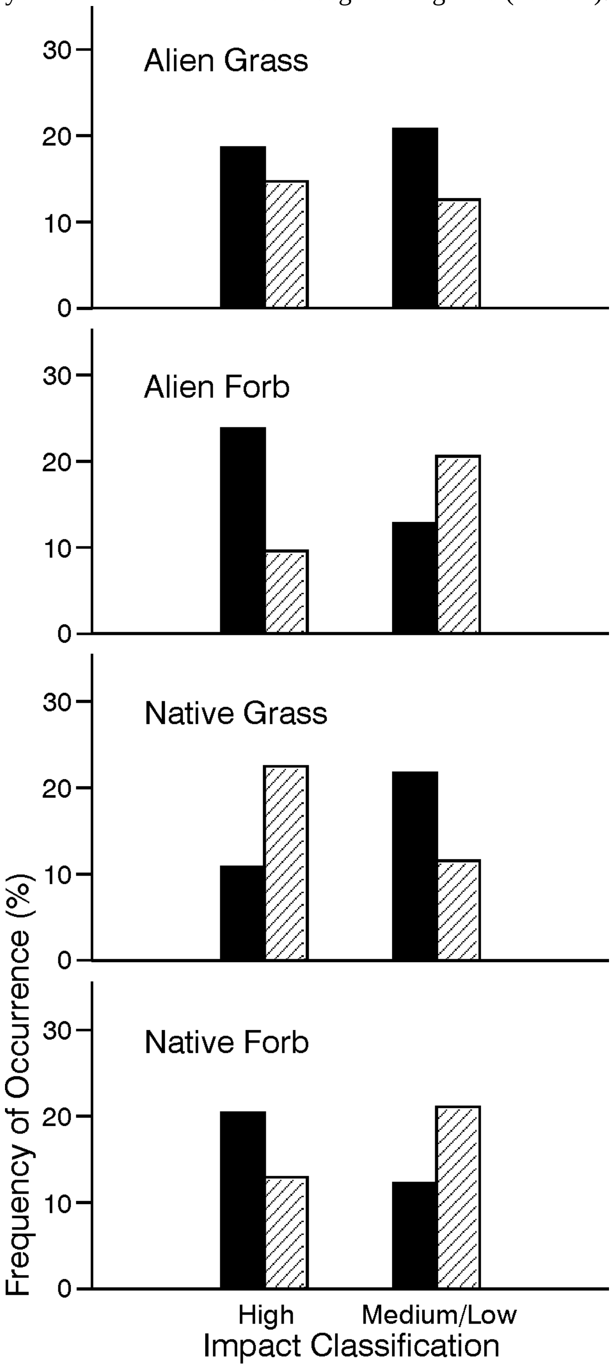


Fig. 2 Frequency of occurrence of species within four herbaceous vegetation guilds between areas of different impact classification from feral sheep on Santa Cruz Island, California, 1984-1993. Percentage of occurrences is the percentage of all occurrences for all species within a guild across years. Black bars represent the period 1984-1989 and hatched bars the period 1991-1993.

The frequency of occurrence of native forbs in high-impact areas was significantly less than alien forbs in all years except 1992 (Table 4). In medium/low-impact areas alien forbs had significantly greater frequency of occurrence than native forbs in all years except 1993 (Table 4). Native grass had the lowest frequency of occurrence of all vegetation categories in the highimpact areas, while in the medium/low-impact areas they had the lowest values in all years except 1984 (Table 4).

From 1984 to 1989 the frequency of occurrence of native and alien forbs was significantly greater in the high impact than medium/low-impact class, but from 1991 to 1993 frequency of occurrence was significantly greater in the medium/low-impact class for both guilds (Fig. 2). In contrast, the frequency of occurrence of native and alien grass was significantly greater in the medium/low-impact class than high-impact class from 1984 to 1989, but from 1991 to 1993 frequency of occurrence was significantly greater in the high-impact class for both guilds (Fig. 2).

There were significant second-order patterns for alpha diversity from 1984 to 1993 (Fig. 3). Species richness ($r = 0.38$, $df = 2,105$, $p < 0.0001$) and species diversity (N2; $r = 0.42$, $df=2,105$, $p < 0.0001$) reached their maximum values from 1985 to 1989, then declined steadily through 1993. Species evenness showed a similar pattern ($r = 0.26$, $df = 2,104$, $p = 0.026$), but mean values for E only ranged from 0.81 and 0.89, so ecologically this pattern was trivial.

Table 5 Multiple regression statistics for the relationship of eight site characteristic variables with species diversity indices and species richness and frequency of occurrence of herbaceous vegetation in grassland plots before and after the eradication of feral sheep from Santa Cruz Island, California, 1984 – 1993. The site characteristic variables were density of feral sheep in the year of sampling (density), the density of feral sheep prior to eradication (pre-density), years post-eradication (ype), % organic litter cover (litter), % vegetation cover (cover), elevation, slope, and aspect. Semipartial correlation coefficient = *Sr*, t-test statistics = t, and probability values = p for multiple regression analyses.

Independent Variable	*Sr*	t	P
Species Richness;			
R=0.54, df=8,99, p<0.0001			
Aspect	.279	3.12	0.002
Elevation	-.454	4.43	<0.0001
Vegetation Cover	-.397	2.16	0.034
Species Diversity (N2);			
R=0.61, df=8,99, p<0.0001			
Aspect	.235	2.79	0.006
Elevation	-.393	4.09	<0.0001
Vegetation Cover	-.645	3.73	<0.0001
Pre-Density	.220	2.19	0.031
Species Evenness (E);			
R=0.52, df=8,97 p<0.0001			
Pre-Density	.235	2.15	0.034
Ype	.426	3.60	0.001
Vegetation Cover	-.614	3.29	0.001
Elevation	-.201	1.91	0.060
Mean Native Forb Frequency;			
R=0.53, df=8,99 p<0.0001			
Pre-Density	.272	2.52	0.013
Slope	.216	2.17	0.032
Native Forb Species Richness;			
R=0.53, df=8,98 p<0.0001			
Pre-Density	.223	2.07	0.041
Slope	.249	2.47	0.015
Elevation	-.269	2.60	0.011
Vegetation Cover	-.355	1.92	0.058
Aspect	.161	1.78	0.079
Mean Native Grass Frequency;			
R=0.51, df=8,99 p<0.0001			
Elevation	-.462	4.43	<0.0001
Litter	-.313	1.92	0.058
Native Grass Species Richness;			
R=0.51, df=8,98 p<0.0001			
Slope	-.264	2.60	0.011
Elevation	-.384	3.67	<0.0001
Vegetation Cover	-.437	2.28	0.025
Mean Alien Forb Frequency;			
R=0.42, df=8,99 p=0.011			
Ype	.251	2.02	0.046
Slope	-.244	2.29	0.024
Elevation	-.222	2.01	0.047
Vegetation Cover	-.397	2.00	0.048
Alien Forb Species Richness;			
R=0.48, df=8,98, p=0.001			
Aspect	.254	2.70	0.008
Elevation	-.436	4.08	<0.0001
Pre-Density	.205	1.84	0.069
Mean Alien Grass Frequency;			
R=0.54, df=8,98 p<0.0001			
Elevation	.535	5.19	<0.0001
Litter	.462	2.89	0.005
Vegetation Cover	.578	3.14	0.002
Alien Grass Species Richness;			
R=0.43, df=8,99 p<0.007			
Ype	-.251	2.03	0.045
Elevation	-.195	1.78	0.077

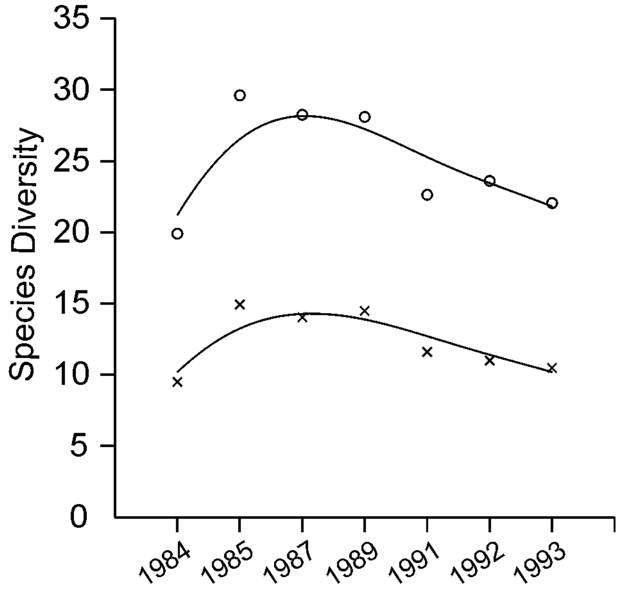


Fig. 3 Species richness (S; shown as open circles) and diversity (N2; shown as X) of herbaceous vegetation during and after eradication of feral sheep from Santa Cruz Island, California, 1984-1993. The eradication programme occurred from 1981-1988.

Species richness was negatively correlated with two site variables and positively correlated with one site variable (Table 5). Species richness increased on low, relatively dry slopes with lower values of vegetation cover. Species diversity (N2) showed the same pattern as species richness, but was also positively correlated with higher sheep density prior to eradication (Table 5). Species evenness was positively correlated with higher sheep density prior to eradication and the number of years after eradication, and negatively correlated with vegetation cover and elevation (Table 5).

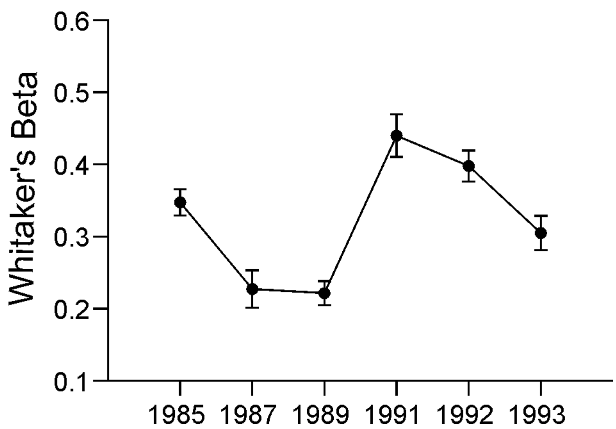


Fig. 4 Turnover of species (Whitaker's beta) of herbaceous vegetation during and after eradication of feral sheep from Santa Cruz Island, California, 1984-1993. The eradication programme occurred from 1981-1988. Error bars are +/- one standard error of the mean.

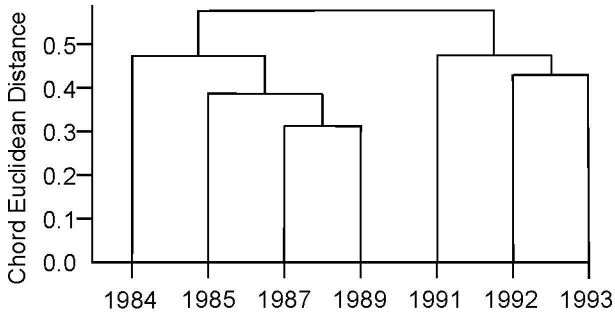


Fig. 5 Dendrogram of similarity (chord Euclidean distance) of herbaceous vegetation during and after eradication of feral sheep from Santa Cruz Island, California, 1984-1993. The eradication programme occurred from 1981-1988. Average linkage was used to construct the dendrogram.

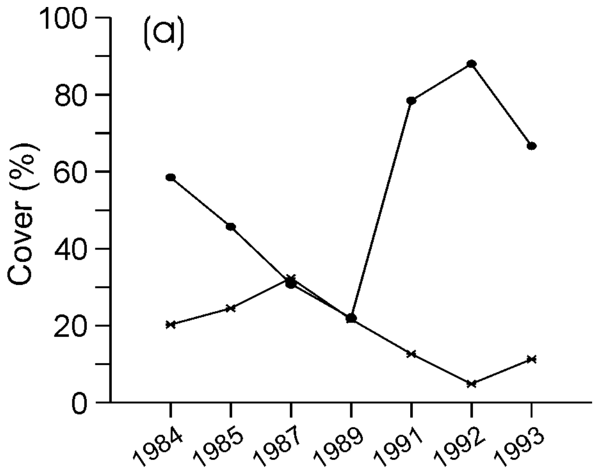
There was a weak negative correlation between species richness and rainfall (r = 0.25, df=1,106, p = 0.010), and species diversity (N2) and rainfall (r = 0.31, df = 1,106, p = 0.001). There was no correlation between species evenness and rainfall.

The turnover of species within plots was greatest between 1984 and 1985, and between 1989 and 1991 (Fig. 4). Cluster analysis indicated that there were two periods with the greatest difference in species composition: 1984 to 1989, and 1991 to 1993 (Fig. 5). The mean coefficient of similarity for 1984-1989 was 0.90, and for 1991-1993 was 0.88. The mean coefficient of similarity between the years from 1984 to 1989 and the years from 1991 to 1993 was 0.81. There was a significant difference among years in the rank order of the 10 most abundant species (Kendall Coefficient of Concordance = 0.264, df = 6, p = 0.015). Two species were primarily responsible for the change in rank abundance. The alien grass *Vulpia myuros* C.C. Gmel declined in rank abundance from 1989 to 1992, then increased in 1993. The native forb *Eremocarpus setigerus* Benth. declined in rank abundance after 1991. Nine of the 10 species with the greatest frequency of occurrence were nonnative.

The percentage cover of herbaceous vegetation did not linearly increase between 1984 and 1993 (Fig. 6a). However, there was a positive correlation between cover and rainfall (r = 0.54, df = 1,106, p <0.0001) (Fig. 6b), so mean cover in the period from 1991 to 1993 was 37% greater than the period from 1984 to 1989. There was a significant negative correlation between rainfall and S (r = 0.25, df = 1,106, p = 0.010) and N2 (r = 0.31, df = 1,106, p = 0.005). Beta diversity had a marginally significant positive relationship with rainfall (r = 0.20, df = 1,106, p = 0.061).

Five environmental variables were retained for the Canonical Correspondence Analysis (CCA); the density of sheep in an area prior to eradication, slope, aspect, elevation, and cover of vegetation. Sheep density in the year of sampling and litter cover did not contribute significantly to the CCA, and years post-eradication and bare ground were not included because of their strong correlation with vegetation cover (r>0.73). The first two CCA axes had clear ecological interpretation, and indicated that distribution and abundance patterns of herbaceous species in grasslands were correlated with topographic features as well as sheep impacts. The first CCA axis was a sheep impact and elevation gradient, with species distribution and abundance patterns varying from high elevation areas with low vegetation cover to low elevation areas with higher vegetation cover (Fig. 7). The second axis was a topography gradient, with slope and aspect having the primary influence on species composition (Fig. 7). The first axis accounted for >50% of the variation in the species abundance data, and the second axis for 22% of the variation in the species abundance data.

There was a significant positive correlation of the mean frequency score of native forbs with slope and density of sheep in an area prior to eradication (Table 5). Species richness of native forbs had significant positive correlation with slope, elevation, and density of sheep in an area prior to eradication (Table 5). Species richness of native forbs also had a marginally significant negative relationship with percentage cover of vegetation and a marginally significant positive relationship with aspect (Table 5). The mean frequency of native grass had significant negative correlation with litter and elevation, while species richness of native grass had significant negative relationships with slope, elevation and the percentage cover of vegetation (Table 5).



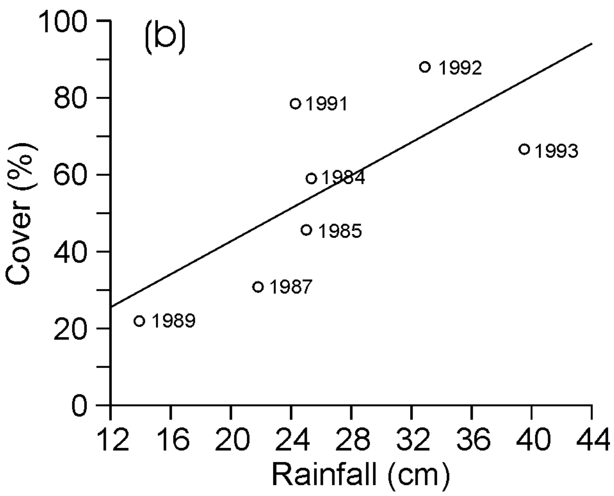


Fig. 6 (a) Change in percentage cover of bare ground (shown as X) and herbaceous vegetation (shown as closed circle) and, (b) the relationship between rainfall and vegetation during and after eradication of feral sheep from Santa Cruz Island, California, 1984-1993. The eradication programme was from 1981-1988.

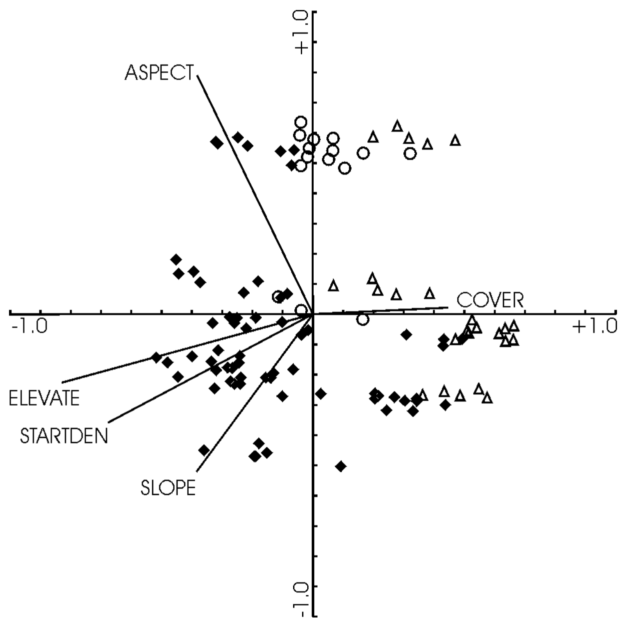


Fig. 7 Canonical Correspondence Analysis biplot of the relationship between five environmental variables and plant species composition in plots with different impacts from feral sheep on Santa Cruz Island, California, 1984-1993. Solid black diamonds are plots categorised as high impact areas, open circles are plots categorised as medium impact areas, and open triangles are plots categorised as low impact areas. Startden is the density of sheep prior to eradication and Cover is percent total herbaceous vegetation cover. Aspect, Elevate, and Slope are geographic attributes of the plots.

The mean frequency of alien forbs had significant negative correlation with slope, elevation, and percentage cover of vegetation, and a significant positive correlation with the number of years post-eradication (Table 5). Species richness of alien forbs had a significant positive correlation with aspect and a significant negative correlation with elevation (Table 5). The density of sheep in an area prior to eradication had a marginally significant positive correlation with species richness of alien forbs (Table 5).

There was a significant positive correlation between the mean frequency of alien grass and elevation, litter, and vegetation cover (Table 5). Species richness of alien grass had a significant negative correlation with the number of years post eradication and a marginally significant negative correlation with elevation (Table 5).

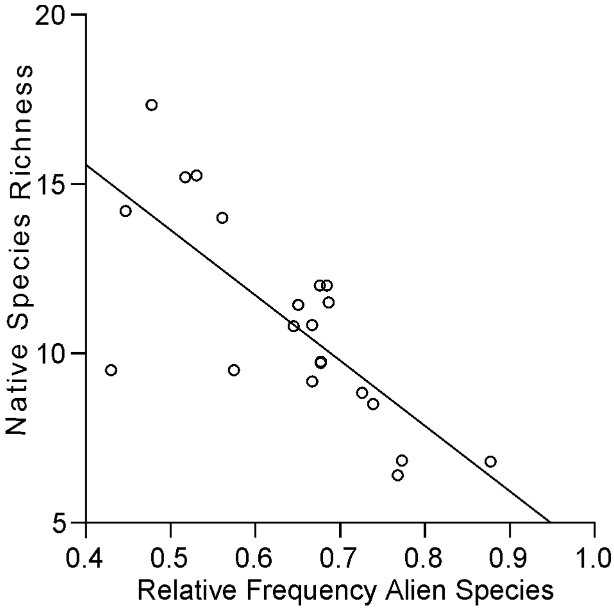


Fig. 8 The relationship between species richness of native grasses and forbs and the relative frequency of alien grasses and forbs during and after eradication of feral sheep from Santa Cruz Island, California, 1984-1993. The eradication programme was from 1981-1988.

There was no significant relationship between species richness of native and alien species. However, there was a significant negative correlation between native species and the mean frequency of aliens ($r = 0.745$, $df = 1,19$, $p < 0.0001$) (Fig. 8).

Endemic Species

The distribution of endemic species increased from 47.5% of the km^2 units during the 1991-1992 survey period to 52.5% in the 1995-1996 survey period (X^2=6.66, df=1, p=0.01). Of the 43 endemic species, 33 were observed in more km^2 units during the 1995-1996 survey. Four of the species occurred less frequently in the 1995 – 1996 survey than during the 1991-1992 survey, and two species, *Malacothrix squalida* Greene (Asteraceae) and *Erysimum insulare* Greene (Brassicaceae) were not observed in either of the survey periods (Table 6).

Twenty-four of the 250 species (9.6%) recorded on the transects from 1991-1995 were endemic. There was a significant linear increase in the percentage of plots that nine of the species occurred in, and none decreased in distribution or abundance. The six species that had a significant linear increase in density were all shrubs (Table 6).

Of the five endemic species that were monitored most intensively, the abundance of one species remained unchanged, two increased in population size, and two had serious declines. No new populations of *Berberis pinnata* were found, and the populations that were known from previous surveys showed no indication that they were expanding their range or increasing in abundance.

Table 6 Change in distribution of 43 endemic vascular plant species between 1991 and 1995 on Santa Cruz Island, California. Surveys are for 190 x 1 km^2 units checked in two periods: 1991-1992 and 1995-1996. Transects are for one hundred 30m x 2m transects surveyed annually from 1991–1995. In the Survey column the symbols '+', '0', and '-' indicate an increase, no change, or decrease (respectively) in the number of units the species was observed in between survey periods. In the Transect column: the symbols '+', '0', and '-' indicate a linear increase, no change, or decrease in the percentage of transects the species occurred in; an '*' indicates that that species had a significant linear increase in density and/or cover; no symbol indicates that the species was not observed in the transects.

Species	Surveys	Transects
Arabis hoffmannii	+	
Arctostaphylos insularis	+	0*
Arctostaphylos tomentosa subsp. *insulicola*	+	0
Arctostaphylos viridissima	+	0
Astragalus miguelensis	+	
Berberis pinnata insularis	+	
Calystegia macrostegia subsp. *macrostegia*	-	0
Castilleja lanata subsp. *hololeuca*	+	+
Ceanothus arboreus	+	0
Ceanothus megacarpus subsp. *insularis*	+	+*
Chorizanthe wheeleri	+	0
Dendromecon rigida subsp. *harfordii*	+	
Dudleya candelabrum	+	
Dudleya greenei	+	
Dudleya nesiotica	0	
Eriogonum arborescens	+	0*
Eriogonum grande subsp. *grande*	+	0*
Eriogonum grande subsp. *rubescens*	+	
Erysimum insulare	0	
Escholzia ramosa	+	
Galium angustifolium subsp. *foliosum*	+	0
Galium buxifolium	-	
Galium nuttallii subsp. *insulare*	+	0
Gilia nevinii	-	0
Hazardia detonsa	+	
Helianthemum greenei	+	0
Hemizonia clementina	+	
Heuchera maxima	+	+
Jepsonia malvifolia	+	+
Lotus argophyllus subsp. *niveus*	+	
Lotus dendroideus subsp. *dendroideus*	+	+*
Lyonothamnus floribundus subsp. *aspleniifolius*	0	
Malacothamnus fasciculatus subsp. *nesioticus*	+	
Malacothrix indecora	0	
Malacothrix saxatilis subsp. *implicata*	+	+
Malacothrix squalida	0	
Mimulus flemingii	+	+*
Quercus pacifica	+	0
Quercus tomentella	0	+
Rhamnus pirifolia	+	0
Ribes thacherianum	+	0
Solanum clokeyi	+	+
Thysanocarpus conchuliferus	-	

One new population of *Arabis hoffmannii* was discovered, but two of the known populations were severely impacted by feral pigs. In 1993, rooting by feral pigs reduced the size of the largest population from 32 stems to 19 stems. The population rebounded the next year to 78 stems, and reached a maximum of 110 stems in 1996. In 1997 the population decreased to 98 stems, but flower and seed production remained high. Intense hunting in the surrounding area eliminated feral pig impacts to the population. In 1995 feral pigs destroyed a second population on an isolated cliff on the north side of the island. This population had been comprised of 15 stems, and no recruitment was occurring. No stems were seen after 1995. The third population fluctuated between 11 and 81 stems from its discovery in 1995 and the last year of surveys in 1998.

Two previously-unknown populations of *Malacothamnus fasciculatus* subsp. *nesioticus* were found. A population discovered in 1992 increased annually from 16 to 27 stems through 1997, and a population discovered in 1997 was comprised of 87 stems and appeared to have been increasing for a number of years. The number of stems from the previously-known population increased annually from six in 1991 to 48 in 1997.

The two endemic species that declined were *Dudleya nesiotica* and *Thysanocarpus conchuliferus*. The density of *D. nesiotica* initially increased between 1991 and 1993, but it declined from a mean of 30.6 stems/m^2 in 1993 to 3.6 stems/m^2 in 1998. The decline was strongly correlated with habitat alteration as a result of an increase in cover of alien grasses and forbs and a related buildup of organic litter (R. Klinger, *in review*). *Thysanocarpus conchuliferus* was found at 11 of the 14 historically-known populations in 1991. In the ensuing seven years only one of these populations continued to persist. Feral pigs had rooted five of the sites, and the habitat at all locations had been heavily modified by an increase in cover of alien annual grass. There were <56 individuals at the remaining known population.

DISCUSSION

It is often assumed that removing nonnative grazers from islands will lead to recovery of native species (Temple 1990; Halvorson 1992, 1994). This assumption can be justified to a certain degree (Ritchie 1970; Meurk 1982; Wodzicki and Wright 1984), but as a general expectation it is probably overly simplistic (Cunningham 1970; Usher 1989; Maunder *et al*. 1998). As the patterns showed on SCI, removing feral animals from islands will lead to a range of complex effects, many of which will be beneficial to native species and many of which may not.

The net effect on species diversity following the eradication of feral sheep from SCI has been relatively minor. Species richness, diversity, and evenness were all related to site characteristics other than just sheep impact. The increase in herbaceous cover and decrease in bare ground that occurred after the sheep were eradicated actually resulted in lower levels of alpha diversity, at least within grasslands. Indeed, there was some indication that alpha diversity might be decreasing as a result of the dominance of alien grasses.

Turnover of species within plots appeared to be most related to the end of the drought and the elimination of the sheep. Changes in turnover at local scales will likely be small, and more a result of changes in the relative abundance of species than changes in species richness itself. Even years that were most dissimilar in composition shared about 85% of the species. Island wide, beta diversity will not be likely to change unless there are major invasion or extinction events on SCI, which seems unlikely.

Reflecting the lack of major changes in alpha diversity, there was no unambiguous change in gamma diversity. Twenty-two species of vascular plants that were never previously recorded on SCI or were considered extirpated have been found since 1987, and no species have been known to go extinct since the sheep were eradicated. It is important to note that 14 of these species that have been recently found are alien (S. Junak, Santa Barbara Botanic Garden, pers. comm.). But, it is unknown whether these species represent new invasions or had simply been on the island and then increased in abundance to detectable levels after the sheep were eradicated.

The relative abundance of native species in grasslands on SCI was determined primarily by an interaction between grazing intensity, competition with alien species, rainfall patterns, and site characteristics. All of these factors had effects on guilds of herbaceous species, and site characteristics had strong effects at the species level. Other studies on the relationship between grazing pressure, species composition, and species diversity have reported similar patterns (Noy-Meir *et al.* 1989; Cornelius and Schultka 1997; Oliva *et al.* 1998).

Rainfall was an important factor affecting the increase of vegetation cover, but the increase in cover did not lead to an increase in the number or relative frequency of native species. Surprisingly, relatively more native species occurred in open areas where sheep impacts had been severe, albeit at low abundance. These tended to be higher and steeper sites with lower frequency of alien annual grass.

Species composition in grasslands was dominated by alien species in all years, and alien annual grasses were the dominant group in all areas, regardless of site characteristics or rainfall patterns. But they were less abundant in high impact areas than low and medium-impact areas prior to the completion of eradication, which in all likelihood was a result of heavy grazing by the sheep. In the five years after completion of the eradication programme and the end of the drought, alien grasses increased in frequency in the high-impact areas relative to the medium and low-impact areas. This pattern has been noted in other studies on SCI. Klinger and Messer (in press) reported a strong correlation between rainfall and cover of alien annual grass in grassland areas where prescribed burns had been conducted.

Both native and alien forbs had greater relative frequency of occurrence in high impact areas than medium and low-impact areas prior to and just after completion of the eradication programme. In the wetter years after the end of the eradication, they decreased in frequency in the high-impact areas, which reflected the increase in frequency of the alien grasses in these areas.

The frequency of native grasses was lower in areas of high impact than medium and low-impact areas prior to the end of the eradication programme. After the end of the eradication programme the frequency of native grasses increased in the high-impact areas. Studies on the relationship between native grasses and grazing in California have noted that they are intolerant of continuous, high intensity grazing (Heady 1977; Bartolome and Gemmill 1981; Mack 1989; Dyer *et al*. 1996). When grazing pressure was removed, native grasses on SCI increased in frequency. But frequency of native grass decreased with increasing amounts of litter, and build-up of litter is characteristic of grasslands in California dominated by non-native annual grasses.

Native species were not displaced by new non-native species invading an area, but by an increase in cover from nonnative species already present in an area. There was no relationship between species richness of alien species and species richness of native species, but the number of native species tended to be lower in areas where the relative frequency of aliens was high. This pattern probably reflects the historical effect of sheep grazing; the levels of species diversity, composition, and cover had been established decades earlier, and ongoing grazing did not change them in any significant manner. After the sheep were removed and environmental conditions became favourable (increased rainfall) the vegetation had a chance to recover. Annual grasses increased most rapidly, especially in the high-impact areas, resulting in the displacement of native species.

An evaluation of how the eradication affected diversity patterns, species composition, and vegetation structure in communities other than grasslands must be made cautiously. Although grasslands comprise almost half the area of SCI, there are several lines of evidence indicating both shrub communities and shrub species are recovering. A study of the Bishop pine *Pinus muricata* community on the north side of the island showed that diversity has increased since the eradication (Wehtje 1994). An analysis of diversity patterns from the 100 transects located throughout the island also showed that levels of alpha and beta diversity in communities dominated by shrubs and trees are greater than in grasslands, and a relatively high proportion of this diversity is comprised of native species (Klinger unpubl. data). There has been no decline in abundance of woody endemic species, and a number of endemic shrubs increased both in distribution and abundance.

Regardless of whether it was dominated by alien species or not, the increase in cover undoubtedly reduced erosion and helped restore natural hydrologic regimes. Ongoing degradation of these ecosystem processes would have made future restoration projects more challenging and expensive, if not impossible (Maunder *et al*. 1998). Besides reducing soil loss and improving watershed quality, another desirable outcome since completion of the eradication programme was the overall increase in distribution and abundance of endemic species. This was long considered one of the most important reasons for eradicating sheep from SCI (Hochberg *et al.* 1980), and in most cases appears to have had a high payoff. It is interesting to note that of the 23 species that showed a positive response to the removal, 20 were woody shrubs or trees. This may be due in part to them being less prone to competitive effects of alien species, most of which are herbaceous. A number of woody alien species occur on SCI, but they tend not to be particularly invasive or still only occur in very small, discrete patches (Junak *et al.* 1995; Klinger unpubl. data).

However, it is telling that the two endemic species that showed negative responses to the eradication were two of the rarest species. Although their impact may be less obvious, alien plants can have many of the same detrimental effects to natural communities as feral animals (D'Antonio and Vitousek 1992; Halvorson 1994). This certainly appeared to be the case with both *Dudleya nesiotica* and *Thysanocarpus conchuliferus*. Both were apparently being affected negatively by alien annual grass, and while a well designed habitat management programme could improve conditions for *Dudleya nesiotica*, an *ex situ* propagation programme will probably be required to preserve *Thysanocarpus conchuliferus*. While it is not unlikely that other populations of *Thysanocarpus conchuliferus* will be found on SCI, it is clear that it is in a precarious demographic position.

In summary, there was no single, consistent community-wide pattern that manifested itself in grasslands on SCI once grazing pressure was eliminated. Diversity patterns and species composition following the eradication of feral sheep were determined by complex interactions between rainfall, topography, and the historical severity of grazing. Alien annual grasses tended to dominate most grassland areas, but this was not a result of new invasions but rather by proliferation of species already occurring in the community. Endemic species as a group appeared to show a favourable response to eradication of the sheep, but declines in at least some of the endemic species were associated with the increase in cover of alien annual grasses. Ten years ago it may have been possible to argue that these patterns and others observed on SCI following eradication of grazers (Brenton and Klinger 1994) were isolated events. But ecologists are developing a greater understanding of ecosystem responses to control and eradication programmes (Zavaleta *et al.* 2001, Zavaleta 2002), and it appears likely that the patterns on SCI are representative of the complex responses that can be expected to occur on both island and mainland areas.

CONSERVATION IMPLICATIONS

It is important to recognise that eradication programmes should only be considered a first step for protecting and restoring native species diversity. It is likely that environmental factors influencing the response of plant and animal species to eradication programmes will vary unpredictably, resulting in some succession patterns that are relatively undesirable for native species. Management activities will likely be needed to prevent unwanted outcomes from eradicating feral animals, such as the displacement of native herbaceous species by alien grasses and forbs that has occurred on Santa Cruz Island.

The necessity of eradication programmes will usually be obvious, but it is critical that conservation scientists and land managers understand the likely outcome of these programmes. Simplistic hopes must be replaced with realistic expectations that many outcomes will be unpredictable, and some will be undesirable. The important strategy will be to try to predict the types of undesirable outcomes that may occur, and while it may be unrealistic to predict exactly what species will be involved, resources and plans can be developed for beginning the process of mitigating these events.

Schuyler (1993) noted that four processes needed to be monitored to document how the Santa Cruz Island ecosystem responded to the removal of feral sheep: (1) changes in vertebrate populations, (2) changes in alien plants, (3) changes in hydrologic regimes, and, (4) changes in erosion processes and soil formation. Of these, only changes in alien herbaceous plants were adequately monitored during the Santa Cruz Island sheep eradication. If the other processes had been monitored as Schuyler (1993) suggested, a more comprehensive evaluation of the ecosystem's response to the eradication could have been made.

Feral animal eradication programmes are underway or planned on many islands throughout the world. By designing extensive monitoring protocols as an integral part of any eradication programme, conservation scientists will be able to better react to some of the unwanted outcomes that will inevitably occur after completion of an eradication programme. A number of different ecosystem parameters should be monitored; monitoring should be initiated before eradication begins; and sampling should continue consistently throughout the eradication phase and at least several years beyond.

It has only been 15 years since sheep were eradicated on the western 90% of SCI, and response from the effects of overgrazing will be ongoing for decades. Succession patterns will vary among communities, and some of those patterns will favour native species. Nevertheless, situations like those of *Dudleya nesiotica* and *Thysanocarpus conchuliferus* are reminders that our lack of understanding of how ecosystems function can lead to unintended outcomes that imperil the species and communities we are trying to protect and restore. Ultimately, we will not judge the success of eradication programmes in terms of the number of nonnative organisms we destroy, but rather the number of native species whose populations we preserve.

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Appendix 1 Estimated sheep density (#/km²) on Santa Cruz Island, California. After 1987 no sheep occurred on the western 90% of the island.

Pasture	1982	1983	1984	1985	1986	1987
Light Impact						
La Punta	26.0	0.0	0.0	0.0	0.0	0.0
Cabrillo	43.5	43.5	43.5	0.0	0.0	0.0
Portezuela	12.0	12.0	12.0	12.0	0.9	0.9
Medium Impact						
Alberts	64.2	5.7	14.2	0.0	2.2	4.7
Pozo/Sauces	131.8	131.8	131.8	131.8	76.5	0.4
Heavy Impact						
Dos Cuevas	200.9	74.7	3.2	1.2	0.0	0.0
North Shore	287.6	198.3	198.3	5.5	0.9	1.7
Laguna	291.2	291.2	291.2	291.2	225.8	1.7
Willows	265.6	265.6	265.6	265.6	199.5	1.1

Alien plant and animal control and aspects of ecological restoration in a small 'mainland island': Wenderholm Regional Park, New Zealand.

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Abstract Since 1965, ecosystem-focused ecological restoration has been undertaken in a small (60 ha) mainland island at Wenderholm Regional Park (134 ha), which lies on a peninsula on the east coast north of Auckland. A 60 ha coastal forest has been fenced to exclude livestock and retired pastureland has been reforested. Brushtail possums *(Trichosurus vulpecula)* and rats *(Rattus rattus* and *R. norvegicus)* have been reduced to very low densities and feral cats *(Felis catus)* and mustelids *(Mustela* spp.) have been controlled. Forest health and New Zealand pigeon *(Hemiphaga novaeseelandiae)* breeding success have improved and large invertebrates are now more abundant. By early 1999, Wenderholm was deemed suitable for experimental releases of birds, which had become locally extinct on the northern North Island mainland. In the first release, 21 North Island robins *(Petroica australis)* were translocated from nearby Tiritiri Matangi Island in March 1999. During the past two years, survival of site-attached robins has been high and they have fledged 46 young, thus the robins have been useful indicators of successful control of some invasive alien mammals. However, despite high productivity, recruitment has been insufficient to compensate for adult losses. Poor recruitment may be due to high rates of juvenile dispersal from the mainland island because of its small size. Linkages with nearby forest areas allow robins to disperse easily, and the ultimate success of the translocation is therefore uncertain. The dispersal distances of species intended for release need to be taken into account in the planning of any new mainland island.

Keywords Revegetation; New Zealand robin; translocation.

INTRODUCTION

Most unmanaged forest reserves on the New Zealand mainland harbour many of the invasive alien mammals introduced by Europeans (King 1990), as well as an increasing number of invasive alien plants (Esler 1987). These introduced species threaten much of New Zealand's remaining biodiversity (King 1984; Esler 1988; Wright and Cameron 1990; Saunders and Norton 2001). Conservation managers in New Zealand have achieved considerable success in removing a range of exotic mammals from offshore islands (Veitch and Bell 1990). During the past decade, the experience gained in removal of invasive species from islands has been applied successfully at some mainland sites (Saunders 1990, Innes *et al*. 1995, 1999, Saunders and Norton 2001). Mainland sites provide opportunities to restore communities which do not occur on islands, or to restore very large areas (Saunders and Norton 2001).

Intensive control of alien plants and mammals has been undertaken at Wenderholm Regional Park (hereafter referred to as Wenderholm, Fig. 1) since 1992, thus it is one of New Zealand's longer running so-called 'mainland islands' (see Saunders 1990, Saunders and Norton 2001). Invasive alien mammals which have occurred at Wenderholm include brushtail possums *(Trichosurus vulpecula)*, ship *(Rattus rattus)* and Norway *(R. norvegicus)* rats, house mice *(Mus musculus)*, rabbits *(Oryctolagus cuniculus)*, hedgehogs *(Erinaceus europaeus)*, feral cats *(Felis catus)*, stoats *(Mustela erminea)*, ferrets *(M. furo)*, weasels *(M. nivalis)* and red deer *(Cervus elaphus)*. Invasive alien plants, which threaten the native forest, include climbing asparagus *(Asparagus scandens)*, kahili ginger *(Hedychium gardnerianum)*, cotoneaster *(Cotoneaster glaucophyllus)*, tree privet *(Ligustrum lucidum)*, woolly nightshade *(Solanum mauritianum)*, periwinkle *(Vinca major)* and kikuyu grass *(Pennisetum clandestinum)*.

When Wenderholm became a park in 1965, a 60 ha coastal forest remnant on a headland was fenced to exclude livestock (Auckland Regional Council 1995). Intensive pest control, especially of possums, did not begin until 1982, and since 1990, invasive alien plants have been controlled.

A study of New Zealand pigeons *(Hemiphaga novaeseelandiae)* which began in 1988 (Clout *et al.* 1995a) showed low breeding success as a result of nest predation, probably mainly by ship rats. As a result, annual rat poisoning began in 1992. Rat and possum control was followed by significantly higher pigeon breeding success (Clout *et al.* 1995b, James and Clout 1995), improved forest health (Dijkgraaf 1997), and an increase in the abundance and diversity of some invertebrates (Craddock 1997). Wenderholm was seen as an ideal site for animal pest control, because being a peninsula, and partly separated from surrounding land by a state highway, reinvasion by some mammalian pests is slower.

Since 1995, some pastureland has been planted with early successional species such as manuka *(Leptospermum scoparium)*, kanuka *(Kunzea ericoides)*, karamu *(Coprosma macrocarpa)*, flax *(Phormium tenax)*, and cabbage trees *(Cordyline australis)*. The objectives of this planting are to enlarge the existing forest patch, to provide linkages with surrounding forest remnants, to restore some former wetlands, and to suppress invasive alien plants.

As a result of improved forest health and successful rodent and possum control, Wenderholm was chosen for an experimental release of North Island robins *(Petroica australis longipes)*(Lovegrove 1998). Robins had been locally extinct for at least a century (Oliver 1955; Heather and Robertson 1997). Robins were considered suitable because (see also Armstrong 2000): (1) They have persisted on the mainland despite introduced mammalian predators, (2) although they may coexist with introduced predators, robins are useful indicators of the numbers of some predators (e.g. rats) (Brown 1997; Etheridge and Powlesland 2001), (3) they have broad niche requirements, surviving in shrub lands, forest, and exotic forests, (4) they are relatively sedentary, thus more likely to remain close to the release site (Flack 1978, Lovegrove 1996), (5) they can be trained to take food, which facilitates capture, captive maintenance and post-release monitoring (Armstrong 1995), and (6) they are conspicuous and relatively unafraid of humans and thus ideal for conservation advocacy.

The major objective of the robin release is to establish a viable population, which might be harvested for future translocations. The key question is: can a small mainland island like Wenderholm support a viable population? Robins do occur in smaller habitat areas on islands free of most or all introduced predators: for example, black robins *(P. traversi)* persisted for many years in a 5 ha forest remnant on Little Mangere Island (16 ha) (Butler and Merton 1992); they occurred on Herekopare Island (29 ha) before being exterminated by cats (Fitzgerald and Veitch 1985); robins were successfully introduced to Motuara (40 ha) and Allports (16 ha) Islands (Flack 1978); and the population of 60 on Tiritiri Matangi Island occupies only 13.4 ha of mature forest habitat (Armstrong *et al*. 2000, Armstrong and Ewen in press a).

These self-sustaining populations on small islands contrast with the mainland, where populations seem now to be confined to large areas (>1000 ha) of forest (extrapolated from Bull *et al*. 1985). This distribution mimics that of the tomtit *(Petroica macrocephala)* in Northland (Ogle 1982), where tomtits now occur only in large forest remnants. These distributions presumably mainly result from the impacts on populations of habitat fragmentation and high levels of predation by introduced mammals (Flack and Lloyd 1978, Moors 1983, Atkinson 1985, Brown 1997). Thus the robin release at Wenderholm, and recent translocations to several other small to medium-sized mainland sites (e.g. Boundary Stream (700 ha) (Howard and Christensen 2000), Trounson Kauri Park (450 ha) (Miller 1997), Kakepuku (135 ha) (Hoverd 2000) and Paengaroa (100 ha) (D. P. Armstrong pers. comm.)), may provide an indicative test of how big a mainland island needs to be to support a viable robin population. The key difference between these sites, and large unmanaged forests with natural robin populations on the mainland, is that the release sites all have intensive control of invasive alien mammals. The assumption, even though small numbers of some alien predatory mammals will be present, is that the small mainland islands approximate offshore islands, which are free of these predators. Thus, like some small offshore islands, small mainland islands should be able to support viable robin populations.

In this paper, we describe the control of invasive alien animals and plants at Wenderholm, reforestation, and the release of North Island robins as part of an ecosystem-focussed ecological restoration programme.

STUDY AREA AND METHODS

Study area

Wenderholm Regional Park (134 ha, 36°33'S, 174°43'E) lies on a peninsula on the east coast about 45 km north of Auckland (Fig. 1). The park is bounded by two tidal estuaries and consists of a hilly (up to 140 m) forested headland comprising about 80 ha, a partly-forested spit of con-

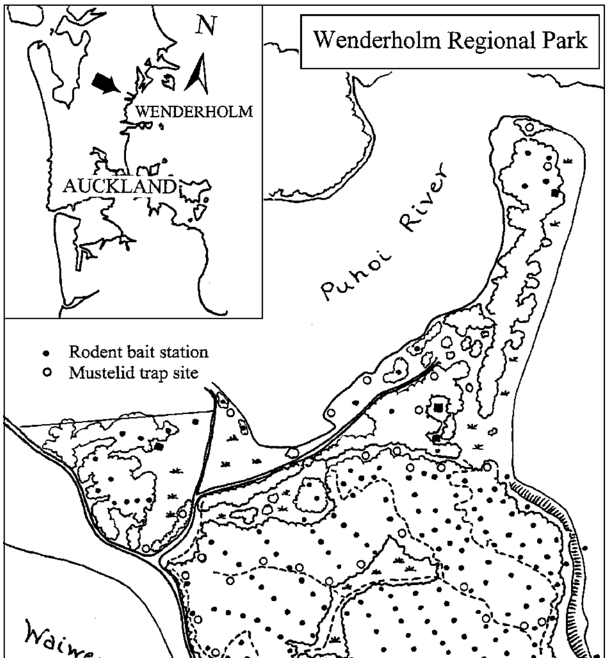

Fig. 1 Wenderholm Regional Park, showing areas of native forest and locations of rodent bait stations and mustelid trap sites.

solidated sands and open coastal pasturelands and wetlands bordering one of the estuaries. A state highway separates part of the park from the inland portion of the peninsula. Before Wenderholm became Auckland's first regional park in 1965, the land was grazed by sheep *(Ovis aries)* and cattle *(Bos taurus)*, and livestock had access to most of the coastal broadleaf forest on the headland. The taraire *(Beilschmiedia tarairi)*-dominated coastal forest on the Wenderholm headland has been identified as having significant values in a regional context (Mitchell *et al.* 1992, Auckland Regional Council 1999).

Livestock fencing and control of alien mammals

During the late 1960s, most of the headland was fenced with standard wooden post and seven-wire farm livestock fences to exclude sheep and cattle from the forest. Between 1982 and 1992, animal pest control concentrated on the control of brushtail possums by poisoning with sodium cyanide (delivered in a concentrated paste bait) and trapping (wire cage and Timms kill-traps). For a two-week period each year, the park was closed to the public to allow possum control using cyanide poison. Recently, other methods of possum control have also been used including Victor No. 1 leg-hold traps and Talon 20P anticoagulant (0.002% brodifacoum) poison. Since the forest area on the headland is quite small, monitoring of possum numbers was undertaken by means of trapping returns and visual assessment of damage to the forest, rather than residual trap catch monitoring, which is more suited to much larger areas (NPCA 2001).

The rodent poison grid (Fig. 1), established in October 1992, consisted of 217 fifty centimetre-long plastic Novacoil drainpipe bait stations placed at 50m x 100m spacings (see Innes *et al*. 1995). Bait stations reduced the risk of non-target poisoning, kept baits dry, and reduced bait wastage. We pinned bait stations to the ground with wire hoops to prevent possums moving them. We poisoned annually from spring to late summer (September to March) with anticoagulant rodenticide. To reduce the risk of selection for toxin resistance, each year we alternated between Talon 50 WB pellets, (active ingredient brodifacoum 50ppm, manufactured by ICI Crop Care, Nelson, N.Z.) and Storm Rodenticide (active ingredient 0.05 g/kg Flocoumafen, manufactured by Shell Agriculture) (Greene *et al.* unpub. data). Each bait station was loaded with six bait pellets at the start of the season and depending on take, topped up at weekly to monthly intervals. In 1999 the drainpipe bait stations were replaced with Philproof rodent bait stations, an improved design with pins to hold the baits in place. From 1999 onwards we used Pestoff Rodent Blocks (active ingredient 0.02 g/kg brodifacoum, manufactured by Animal Control Products Ltd., Wanganui, N.Z.) in these bait stations. These bait stations greatly reduce bait wastage, because rats cannot remove large quantities of bait.

In February 1999, a month before the robins were released, we installed permanent perimeter and central lines of mustelid kill traps (No. 6 Fenns) on the headland (Fig. 1). We spaced 43 double trap sets (see Sim and Saunders 1997) at approximately 100 metre intervals baited with fresh and plastic eggs. We placed the traps in a tunnel of 12 mm galvanised wire bird netting, covered with leaf litter to reduce visibility to non-target species. These traps were checked twice weekly.

We monitored rodent abundance before and after poisoning using a snap trap line consisting of between 25 and 36 stations at 50 m spacings running across the centre of the study area. Each station had two Ezeset rat traps, one baited with peanut butter and the other with cheese. The rat traps were run for three consecutive nights and indices of abundance were calculated according to the method of Nelson and Clark (1973). We also measured rodent abundance indirectly from the rate of removal of baits from the bait stations (Greene *et al*. unpub. data).

We controlled feral cats by shooting and cage trapping as necessary. Rabbits are not significant pests at Wenderholm. They were controlled by night shooting and poisoning with pindone anticoagulant baits, which were broadcast by hand in areas where rabbits were active, during the annual park closure periods.

Reforestation and control of alien plants

We studied areas of seral forest at Wenderholm to determine the proportions of the major species, so that these patterns could be copied in the plantings. We collected seed locally, and the nursery at the Auckland Regional Botanic Gardens grew the plants. Manuka and kanuka were grown in 10 cm x 10 cm peat pots, which could be conveniently stacked onto plastic bread trays, facilitating transport from the nursery to the planting sites. Other species were supplied in PB3 or PB5 bags. In early winter (May-June), the trees (aged 9-18 months) were planted at approximately one-metre spacings. Just before planting, manuka and kanuka were pruned to 0.3 m to encourage root development and reduce wind damage. Planting sites with invasive kikuyu grass were sprayed with 1% glyphosate herbicide up to a year before planting, sometimes more than once, to ensure effective control. The dead grass formed a deep mulch, which protected the young trees from wind and helped retain soil moisture. A sprinkle of slow-release fertiliser was provided for each tree. The plantings received very little aftercare apart from control of woody invasive weeds. We surveyed the park annually and mapped the sites of weed infestations. Weed control was carried out using physical and chemical methods following Veitch and MacArthur (1997).

Robin translocation

A proposal to translocate North Island robins to Wenderholm from nearby Tiritiri Matangi Island was prepared according to guidelines provided by the New Zealand Department of Conservation. The robins on Tiritiri Matangi Island were considered suitable for a translocation to the mainland, because the founding birds in that population were sourced from Mamaku on the North Island mainland in 1992 (Armstrong 1995). The Department of Conservation granted permission to remove up to 30 robins (10 adult males, 10 adult females, and 10 juveniles) from Tiritiri Matangi. Capture, colour banding, captive maintenance, translocation, and monitoring methodologies followed closely those of Armstrong (1995). Monitoring included a systematic survey by up to 15 people along the grid lines each spring.

RESULTS

Livestock fencing and control of alien mammals

Since livestock was removed, the understorey has regenerated strongly. In damper areas the understorey is dominated by supplejack *(Ripogonum scandens)* and matata *(Rhabdothamnus solandri)*. On the seaward and northern slopes, kawakawa *(Macropiper excelsum)*, hangehange *(Geniostoma rupestre)*, *Coprosma rhamnoides*, and *Gahnia* spp. dominate the understorey. In many parts of the forest there has been significant regeneration of nikau *(Rhopalostylis sapida)* as a result of seed dispersal by New Zealand pigeons. Removal of livestock alone has greatly improved the forest for birds, as there is more abundant nectar and fruit in the shrub layer, as well as more foliage to harbour invertebrates.

Possums have been practically eradicated from the park as a result of intensive control since 1990 (Fig. 2). During 2000, a 500 ha buffer area inland from Wenderholm was trapped and poisoned using the methods described above, and possum numbers were reduced to below a 5% residual trap catch index (NPCA 2001; S. Hix pers. comm.).

Results from the rodent index line show that annual poisoning has reduced rat numbers to very low levels in all seasons since poisoning began (Table 1). However, in some years (e.g. 1992-1993, 1995-1996) the mouse population remained high or possibly increased. Bait-take from the bait stations followed a similar pattern in most years with 60-80% of baits taken in the first month of baiting. Thereafter bait take declined to around 15% (Green *et al*. unpub. data). There has been no evidence of rat predation at any robin nest (n=30) in the two breeding seasons to date.

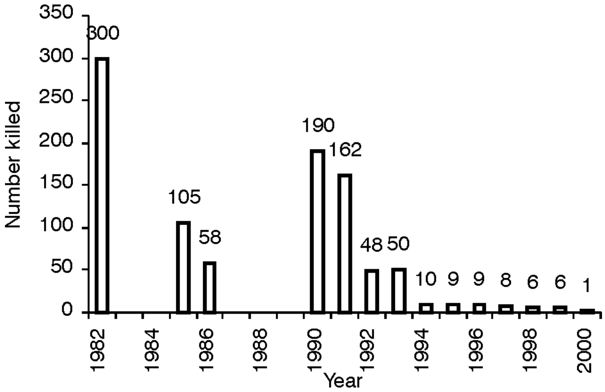

Fig. 2 Number of brushtail possums killed by poisoning and trapping at Wenderholm Regional Park from 1982 to 2000.

Table 1 Rodent indices (number trapped/100 trap nights) at Wenderholm 1990-2000 (after Greene *et al.* unpub. data). Index trap lines in 1990 and 1991 consisted of 72 traps operated for four consecutive nights, data from Clout *et al.* (1995a). Between 1992 and 2000, the index trap line consisted of 66 traps operated for three consecutive nights, and in 2001 it consisted of 50 traps operated for four consecutive nights.

Year	Rat index		Mouse index	
	Pre- [1] season	Post- [2] season	Pre- [1] season	Post- [2] season
1990	10.8		0.0	
1991	15.2		0.0	
1992-93	0.0	0.74	0.0	9.23
1993-94	0.0	0.0	1.49	1.48
1994-95	0.0	0.0	0.0	0.0
1995-96	0.45	0.0	5.84	21.47
1996-97	0.45		5.84	
1997-98	6.0	0.0	12.0	2.29
1998-99	0.52	0.0	2.06	0.0
1999-00	0.0	1.81	1.05	1.81
2000-01	3.37	1.71	5.06	10.85

[1] August, [2] April

Between February 1999 and February 2001, four ferrets, 14 stoats and nine weasels were caught in the Fenn traps. Two cats, one feral (cage-trapped) and the other tame were caught during the period.

Re-forestation and weed control

Between 1995 and 2000, about 28,500 native trees were planted over four hectares of the headland and in a small (0.5 ha) wetland. Four species formed 95% of the plantings, and were used in the following proportions: manuka 43%, kanuka 39%, karamu 7%, and kowhai 6%. The remaining 5% of plants included flax, cabbage tree, kahikatea *(Dacrycarpus dacrydioides)*, mahoe *(Melicytus ramiflorus)*, pohutukawa *(Metrosideros excelsa)*, karo *(Pittosporum crassifolium)*, whau *(Entelea arborescens)*, and puriri *(Vitex lucens)*. By the end of 2000, the oldest trees were already up to five metres tall and the plantings formed a dense shrubland along the forest edges, which has suppressed the invasive kikuyu grass. Intensive weed control, using the methods outlined by Veitch and MacArthur (1997) has greatly reduced infestations of climbing asparagus, kahili ginger, tree privet, and woolly nightshade. These species now require relatively small-scale annual maintenance programmes to keep them in check.

Robin translocation

We translocated 21 robins (nine adult and two juvenile males, seven adult and three juvenile females). Although this number was less than the original target of 30, we considered this sufficient to form the nucleus of a new population. The release site was located close to the seaward end of the reserve, and was chosen to maximise the distance the birds would need to travel to the inland boundary of the park. It was hoped that this strategy would result in most of the birds remaining inside the managed area.

Survival of translocated birds

At the start of the first breeding season, six months after release, at least 13 (61.9%) robins were still alive. These comprised six pairs inside the park, which had settled within 500 m of the release point, and a male, which settled in a contiguous forest area two kilometres outside the park. Some birds that dispersed after the release could have been missed, because apart from forest areas within about two kilometres of the park, other more-distant places were not searched immediately. By September 2000, eight of the founding birds remained, so that year one to year two survival (taken from the systematic spring surveys) was 61.5%. In July 2001, all eight founding birds were still alive.

Results of first two breeding seasons

During the 1999-2000 season, six pairs built 14 nests of which 10 (71.4%) were successful. Twenty-three young fledged (3.8 young per female). Four nests failed for various reasons, but none of these failures were due to rat predation. Four fledglings disappeared before they became independent. Some of these losses may have been due to morepork *(Ninox novaeseelandiae)* predation.

During the 2000-2001 season, five pairs built 16 nests of which 12 (75%) were successful. Twenty-three young fledged (4.6 young per female). Of the four nests that failed, three were abandoned and one was possibly preyed on (unknown predator).

Nests were built in a variety of sites, typical for robins (see Heather and Robertson 1997, Powlesland *et al*. 2000, Armstrong *et al*. 2000). At Wenderholm, nest sites ranged in height from 1-12 metres above the ground, with a mean height above the ground of 5.43 metres (n=30, SD=3.13). Three low nests sited between 1-2 metres were all successful.

Survival and recruitment of young

At least 19 of 23 young that fledged in 1999-2000 and 21 of 23 young that fledged in 2000-2001, are assumed to have become independent. By September 2000 only 4 of 23 of the 1999-2000 young (two males and two females) were known to be alive. Three had settled inside the park, where they had bonded with existing adults, and one had settled outside the park. Assuming that there are no other young birds surviving outside the reserve, recruitment was insufficient to compensate for adult mortality as the population (including birds outside the park) declined from 13 in September 1999 to 12 in September 2000.

DISCUSSION

Livestock fencing and control of alien mammals

After livestock was excluded, the forest developed a dense understorey. In addition to supplying food for birds, there are other benefits such as seclusion from predators for birds and their nests, and shelter. Control of rodents and possums at Wenderholm has been followed by: increased productivity in New Zealand pigeons (Clout *et al*. 1995a; Clout *et al.* 1995b; James and Clout 1995); greatly improved nectar, fruit, and seed production in the forest (Dijkgraaf 1997); greater abundance of invertebrates (Craddock 1997); and successful breeding in translocated robins (this study). Although rodent control using the existing 50m x 100m poison grid will continue, there is some concern about the residual effects of second-generation anticoagulants (Haydock and Eason 1997). Future rodent poisoning at Wenderholm may include changing to a more environmentally acceptable toxin. Annual possum control in the park and the surrounding buffer will continue, so it should be possible to maintain possum numbers permanently at very low levels. Until an effective method is found to remove mustelids, the permanent mustelid trap line will be maintained as a precautionary measure. We do not know whether any robin or robin nest was lost to mustelid predation at Wenderholm. Mustelids, especially stoats, although not as important as rats, are known predators of robin nests (Moors 1983), and stoats are believed to have preyed on some pigeon nests at Wenderholm (Clout *et al.* 1995b; James and Clout 1995).

Reforestation and control of alien plants

Reforestation at Wenderholm has created shrub lands, which should enhance regeneration of the coastal forest, and shade out exotic weeds such as kikuyu grass. Reforestation should be integrated with weed control, because fast-growing canopy-forming species such as kanuka can greatly reduce future weed control costs. As the plantings grow taller, litter is forming, and seedlings of mapou *(Myrsine australis)*, kawakawa, and various sedges *(Carex* spp.) are appearing. Weed control will need to continue, especially of shade-tolerant bird-dispersed species such as climbing asparagus, kahili ginger, and woolly nightshade. Climbing asparagus has the ability to transform forest floor communities by smothering native shrub layer species, and completely covering areas of leaf litter favoured by robins. The annual weed surveys are proving their worth, because incipient infestations of new or known species can be detected and removed before they become a greater problem.

Table 2 Survival of robins at three mainland island release sites in the North Island during the first full year after release. All three sites had similar management regimes to control introduced predatory mammals. Data for Boundary Stream after Howard and Christensen (2000) and Paengaroa (D. P. Armstrong pers. comm.).

Location	No. alive year one	No. alive year two	% survival (±95% confidence limits
Wenderholm	13	8	61.5 (34.5-88.5)
Boundary Stm	24	17	70.8 (51.8-89.5)
Paengaroa	19	12	63.2 (41.2-85.2)

Table 3 Productivity of robins at three mainland island release sites in the North Island during the 1999-2000 breeding season. All three sites had similar management regimes to control introduced predatory mammals. Data for Boundary Stream after Howard and Christensen (2000) and Paengaroa (D. P. Armstrong pers. comm.).

Location	Breeding pairs	Young fledged	Fledged /pair
Wenderholm	6	23	3.8
Boundary Stream	8	20	2.5
Paengaroa	11	16	1.5

Robin translocation

A small population of robins now exists at Wenderholm. Compared with a number of other sites (Armstrong *et al.* 2000; Hoverd, 2000; Howard and Christensen 2000; Powlesland *et al.* 2000), the birds have survived well and bred very successfully (Tables 2 and 3). As an indicator species, the robins show that the levels of nest predation by rats are negligible, suggesting that the rodent poisoning at Wenderholm has been at least as effective as at two other mainland islands (Tables 2 and 3). However, the first year's survival and recruitment figures at Wenderholm show that insufficient young survived to replace losses. It is possible that more young will be recruited in subsequent years, thus it is probably too early to judge whether robins will successfully establish in the park.

However, if recruitment after the 2000-2001 season is again insufficient, a key aspect for future management at Wenderholm will be to enhance the survival of the young after they leave their natal territories. At least one young bird dispersed out of the park, and there were probably others. Mortality of young outside the park is probably high, because apart from possum control, there has been no other predator management specifically targeting rats, cats, and mustelids. The creation of another managed area, where two birds have already established contiguous territories two kilometres outside the park, might allow young produced in the two sites to recruit in both directions, benefiting both populations. At present young dispersing from the park are lost into a large surrounding sink of unmanaged habitat.

Dispersal of robins out of the park will be an ongoing factor at Wenderholm, because forest corridors link the park with other forest areas further inland. If the dispersing birds survive and breed, this is beneficial, as it allows robins to colonise new areas. However, easy dispersal makes establishing a larger core population inside the mainland island at Wenderholm more difficult. While it is possible to establish viable populations on even small offshore islands, mainland islands probably need to reach a minimum effective size (determined by the dispersal distances of the various species being conserved inside them) before they can support viable populations (Ron Moorhouse pers. comm.). This emphasises the need to protect the existing population inside and outside the park either by creating another managed area to encompass the birds outside it or by extending the existing managed area to include them. The aim over the next five years should be to establish a population of at least 20-25 pairs at Wenderholm. Work on Tiritiri Matangi Island (Armstrong and Ewen in press) indicates that a population of this size, given sufficient protection, should be stable in the long term.

At Wenderholm we have assumed that provided mammalian predators are managed effectively, 60 ha of forest will be large enough to sustain a robin population. We also assume that a small protected area will be better than a large unprotected one. This is based on robins surviving in small areas on the various islands mentioned above. However, robins survive on small islands only if they are predator-free: for example they disappeared from Herekopare (29 ha) and Mangere (113 ha) Islands after cats arrived (Fitzgerald and Veitch 1985; Butler and Merton 1992) and from Big South Cape (400 ha) and Solomon (25 ha) Islands after ship rats arrived (Atkinson and Bell 1973). Robins persisted on larger islands with some introduced predators: for example Little Barrier Island (3000 ha) with feral cats and Pacific rats *(Rattus exulans)* (Turbott 1961), and on Kapiti Island (2000 ha) with possums, feral cats, Pacific rats, and Norway rats (Wilkinson and Wilkinson 1952). On the mainland, robins have persisted with the full suite of introduced mammalian predators, but only in very large forest areas. The recent (1997) local extinction of robins at Boundary Stream (700 ha) (reintroduced in 1998 after a mainland island was created), is a good example of an extinction event in a smaller mainland forest block. The current restriction of natural robin populations in the central North Island to larger forest areas parallels the distribution of the tomtit *(Petroica macrocephala)* in Northland (Ogle 1982).

Wenderholm might be a useful testing ground to show just what the lower size limit is for a viable robin population in a small mainland island. If robins fail to establish, then this will be a useful case study, because there are increasing demands from land care groups to release locally-extinct native birds into small privately-owned mainland islands. To study this aspect further, it would be useful to compare translocations of robins to a number of small mainland islands such as Wenderholm with a number of larger mainland islands. However, to separate possible site differences from habitat size, an adequate sample size would be required. Since only a few robin translocations are done each year, such a study in any one year is probably logistically impossible. However, over the next few years, information on releases to large and small sites will accumulate, and this may show some interesting trends. For example during 2001, robins have already been released at several new sites including Mangaokewa (200 ha, H. Speed pers. comm), Karori Sanctuary (250 ha, R. Empson pers. comm.), Waotu (two reserves of 22 ha and 10 ha, G. Stephenson pers. comm.) and Hunua (a 600 ha mainland island within a 15,000 ha forest (T. Lovegrove unpubl. data). Plans are also afoot to release robins at Bushy Park (85 ha, D. Armstrong pers. comm.) and at Mapara (1400 ha, H. Speed pers. comm.) during 2001 and 2002.

The present population of five robin pairs at Wenderholm may be vulnerable to chance factors and therefore be too small to be sustainable; thus a second release might be necessary. However, we suggest that the population should be monitored for at least another year before any more birds are released. Robin populations have successfully established in the past from very small numbers of founders (as few as five) (Flack 1976; Butler and Merton 1992). This is supported by analysis of the establishment of robins on Tiritiri Matangi (Armstrong and Ewen 2001), which suggests that a second release at Wenderholm is probably not too urgent, and much could be learned from further management and monitoring of the existing population.

CONCLUSIONS

Ecological restoration at Wenderholm demonstrates that it is possible to control a wide range of invasive animals and plants in a small mainland island, and to keep populations of certain invasive alien species at low levels. For invasive mammals this has been achieved with an annual pulse of poisoning and trapping rather than continuous control. The local geography (a peninsula) has probably facilitated this by slowing re-invasion by some mammals. There is already evidence that a number of ecological processes have been re-vitalised as shown by improved forest health, increased numbers of invertebrates, and improved productivity of some native bird species. The outcome of an experimental release of North Island robins is still uncertain. While the translocation shows that robins can survive and breed very successfully at Wenderholm, and that they are probably useful indicators of the success of rat control, this 60 ha mainland island could be too small to support a viable population in isolation from surrounding forest areas. The limiting factor may be juvenile dispersal distances, however more information on this is needed. The dispersal distances of species intended for release need to be taken into account in the planning of any new mainland island.

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Eradicating invasive plants: Hard-won lessons for islands

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Abstract The record of eradicating invasive plants, whether on islands or continents, consists of few clear victories, some stalemates, and many defeats. Instructive, if hard-won, lessons have nevertheless been learned. (1) The ideal eradication campaign would see all individuals of a potentially-invasive species destroyed immediately upon their arrival. Few immigrants meet this fate; more usually there is a failure to act until damage has been inflicted by the invader. (2) Failing the destruction of all immigrants upon their entry, maximum effort should be lodged against the immigrants' small, isolated foci. As with (1), implementing this sound advice has often proven difficult. However, the radical reduction of the range of *Striga asiatica* in North Carolina (USA) represents the clearest sustained application of this principle. (3) Eradication or even effective control of invasive species requires repeatedly surveying the same area for surviving plants. Virtual eradication of *Schinus terebinthifolius* and other invasive species on tiny islands in Bermuda has clearly succeeded through such diligence. (4) Control or even eradication of a single invasive species may ultimately produce little benefit, if its demise only sparks the rise of another non-indigenous species (e.g., the role reversals of invasive aquatic macrophytes in southern Florida, following biological control). Islands, with their intrinsic borders and geographic isolation, provide excellent locations for experimentation within which these lessons can be honed, thereby identifying both the effective and ineffective components of any eradication effort.

Keywords nascent foci; *Centaurea trichocephala*; *Salvinia molesta*; invaders; invasion; Bermuda; Hawaii.

INTRODUCTION

From the perspective of their human colonisers, oceanic islands have native floras that did not satisfy human needs or at least their desires in plants (Mack 1999, 2001). This perceived paucity in the native floras of islands has long sparked the transfer of huge numbers of non-indigenous species to islands from widely separated regions, mainly continents (e.g., Streets 1962; Whistler 1991). Islands' lack of biotic diversity compared with continents has also meant that they are poorly protected by biotic barriers to naturalisation and invasion (Mack 1996). They commonly lack native representatives for many of the taxa from which arise the natural enemies potentially capable of extirpating immigrants. For example, Hawaii has no native isopterans; Fiji has no native bruchids, both important plant foragers. The combination of increased propagule pressure and greater intrinsic vulnerability to invaders has meant that oceanic islands have become, in the past 500 years, among the most floristically-transformed land-masses. For example, Bermuda, a British possession since the early 17th century, has long been a transit point and provisioning station between the Western Hemisphere and Europe (Craven 1938). With few native species that were deemed commercially valuable, plant introduction was rampant; by 1918, more than 300 species were listed as naturalised (Britton 1965), all packed onto approximately 50 km^2. Oceanic islands commonly contain several-fold more non-indigenous plant species than an equivalent area of mainland (Lonsdale 1999). In a sense, deliberate eradication (i.e., the total elimination of a species) compensates, however feebly, for the inability of the islands' native biota to provide resistance to plant naturalisation. On the other hand, the possibility of preventing re-infestation is greater on islands, so eradication campaigns may be more fruitful.

The central questions we pose here deal with how eradication of non-indigenous plants can be implemented on islands. In assembling and evaluating the answers, we have drawn on examples from islands and continents, seeking common denominators among both successful and unsuccessful eradication efforts. Best practices in eradication (*sensu* Wittenberg and Cock 2001) are applicable to islands and continents. Furthermore, we have too few well-documented cases to exclude automatically examples based simply on the size of the land-mass in which they occurred. For much the same reason, examples from oceanic islands are applicable to ecological islands (i.e., habitats surrounded by distinctly-different environments), (e.g., lakes and ponds). From the standpoint of evolution, dispersal, biogeography (MacArthur and Wilson 1967) and the efficacy of eradication, these ecological islands are analogous to geographic islands.

Eradication of non-indigenous species anywhere is a quintessential application of science to technology. A clear distillation is needed of simply, 'what has worked' and, 'what has not.' Consequently, we strive here to assemble and discuss relevant examples in order to form basic lessons that can be placed readily into practice.

DEFEATS IN ERADICATING INVASIVE PLANTS: OPPORTUNITIES LOST

Islands, as well as continents, provide ample examples of prime opportunities lost in the eradication of introduced species that later became invaders. Among the least-justifiable episodes was the establishment and spread of the sprawling shrub, *Clidemia hirta* in Hawaii. The well-named "Koster's curse", was first collected on Oahu in 1949 but reputedly detected in Hawaii in 1941 (Anon.

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1954). As early as 1949 it was escaping from gardens. The same population had grown to cover less than 100 ha in 1952. As recently as 1961, Plucknett and Stone included *C. hirta* among "…several species which seem to be spreading in certain areas but which at present cannot be classed as dangerous or even common weeds." This lack of concern about the shrub's future proved unfortunate. By 1977, *C. hirta* had covered about 31,350 ha on Oahu (Wester and Wood 1977). In the most recent range estimation for Oahu of which we are aware, it reportedly occupies 100,000 ha (Smith 1992). Moreover, it has now spread to five other main islands in the state (http://www.hear.org/AlienSpeciesInHawaii/maps/CliHirHI.htm). Even by 1949 the reputation of *C. hirta* should have prompted its vigorous eradication. Koster's Curse had already caused so much damage in Fiji that it was declared a noxious weed by 1920 (Patel 1971). The Hawaiian governmental response in 1954 was not an eradication plan, which was then still possible, but control through release of a thrip, *Liothrips urichi*. This action simply copied a biological control effort begun on Fiji in 1930. Putatively, the thrip prevented the shrub from entering croplands and pastures; it did not, however, prevent its spread into native forest. Given the extent that *C. hirta* has now spread in Hawaii, coupled with its dispersal by birds (Wester and Wood 1977), its eradication seems exceedingly unlikely.

Other species (and the date by which they were first collected) that represent lost opportunities for eradication in Hawaii are *Hypericum perforatum* (1961), *Pistia stratiodes* (1932 or 1933), *Mollugo cerviana* (1975), and *Carduus pycnocephalus* (1986) (Wagner *et al*. 1990). *Olea europaea* subsp. *europaea* was first collected in 1982 (Wagner *et al.* 1990), although it may have entered Hawaii much earlier and remained unrecognised amongst specimens of *O. europaea* subsp. *africana*. Detection of the generalist hemiparasite *Cuscuta campestris* (Parker and Riches 1993) in 1955 and its subsequent naturalisation are particularly galling. Dodders form one of the few genera prohibited from entry into the United States (Westbrooks and Eplee 1999). Because *C. campestris* is native to the U.S. mainland, its arrival in Hawaii did not merit the eradication campaign that would have been triggered if it originated outside the U.S. (Westbrooks and Eplee 1999), a ludicrous gap in federal law. Important here is that by the time each species was first detected and was still confined to a few small populations, its propensity for invasion elsewhere had already been well documented.

EARLY DETECTION AND ERADICATION: OPPORTUNITY GRASPED

Lessons from another arena

The difficulty of detection and eradication of non-indigenous plants is forcefully illustrated by an example from an unexpected quarter – the Counter Cannabis Field Operation of the U.S. Drug Enforcement Administration in Hawaii. Conventional wisdom maintains that if a non-indigenous species is to be eradicated, it must be detected soon after its entry into a new range. Equally well understood, however, is that most non-indigenous species are exceedingly difficult to detect in low numbers. They are partly hidden under native species, or their habit, stature, leaf colour, morphology and texture, or any other visible features are indistinguishable from their native neighbours, except under close examination. Yet so strong is this link between early detection and eradication that much effort has been devoted to finding tools that aid detection. Limitations of the current range of tools to facilitate early detection have been discussed elsewhere (Mack 2000). In what is likely the most concentrated effort in the early detection of a non-indigenous plant species – the surveillance for illicit *Cannabis sativa* in Hawaii – visual detection from the air remains the tool of choice. Field agents of the U.S. Drug Enforcement Administration in Hawaii develop a search image for *C. sativa*. They detect plants primarily from the air in either fixed wing aircraft or helicopters flying at approximately 150 m altitude and usually above dense forest. Eradication either by aerial spraying or destruction of plants by ground teams follows immediately after aerial detection. Despite a level of surveillance and follow-up destruction that has been rarely, if ever, matched in the eradication of invasive species, *C. sativa* continues to be found in rural Hawaii (Anonymous source, Honolulu District Office, U.S. Drug Enforcement Administration).

This account may seem a far cry from the eradication of invasive plants, but we contend that some parallels do exist. Even though *C. sativa* is not even naturalised in Hawaii (Wagner *et al.* 1990), much less invasive, it persists through animal (i.e., human), dispersal into suitable habitat. Thus, its spread is analogous to the spread in Hawaii of other non-indigenous species, such as *Clidemia hirta*, *Miconia calvescens* or *Psidium cattleianum* (Cuddihy and Stone 1990). Despite the apparent inability of *C. sativa* to persist without cultivation in Hawaii, its illicit cultivation ensures that it occurs in small, widely scattered populations. This distribution obviously hampers its detection, also producing a distribution similar to a bird-dispersed non-indigenous species, early in its residence in a new range (e.g., *Miconia calvescens*). Furthermore, the difficulty of early detection for any non-indigenous species beneath a forest canopy is certainly illustrated with illicit *Cannabis* cultivation in Hawaii, in which the growers hide their plots in dense forest. Twenty-four years after initiation of this extensive aerial surveillance and eradication programme, the results represent broad-sense control but hardly eradication. In 1999, 3,413,083 *Cannabis* plants were destroyed in the U.S. (http://www.dea.gov/programs/marijuana.htm); more than half in Hawaii. The result of this initiative has caused a reduction in the amount of *Cannabis* (or the comparative ease of its detection) in the field compared with results in the past. The species is too widespread, has a long-lived repository for propagules for re-infestation (as long-lived as the human desire to take illicit drugs), and is too readily dispersed for eradication to be a realistic goal. It is worrying that many invasive species in Hawaii fit this template.

Plant eradication: a few slender victories

Can eradication of potentially-invasive plants ever be achieved? In citing the record for invasive animal eradications in Florida, Simberloff (1997) found no analogous list of successful pest plant eradication programmes for the United States or elsewhere. A list can indeed be assembled, although it is a slender one so far. These examples are instructive because they share common features – they consistently involve very small plant populations, usually a few hundred individuals, comprising one or only a few foci. Detection was apparently very early in the immigration; the decision to destroy all the immigrants was swift, and most important – repeated field operations at least reduced the non-indigenous species below levels of detection, even if extirpation cannot always be demonstrated.

Eupatorium serotinum is a herb native to North America. It was first detected in Australia in 1962 at Nerang, Queensland where it was growing in the cattle salesyard on a railway property. The population grew from occupying 9 m^2, to occupying about 49 m^2 just a year later. Although its identity was not confirmed until September 1963, authorities from the Queensland Department of Lands began to destroy the population four months earlier. This action was timely because some plants were already producing abundant seeds. The site was surveyed annually until 1980, and specimens were routinely collected (e.g., A. J. Tomley (s.n.), Alan Fletcher Research Station, Brisbane, Queensland). Any newly emergent plants were hand-pulled, and by 1980 no additional plants had been found for several consecutive years (corres. of K. L. Kay, Biological Branch, Queensland Department of Lands, 1980).

Similar conscientiousness in eradication was also displayed in Queensland's Mt. Tarampa District upon first detection in 1967 of several small populations of *Helenium amarum*, another composite herb native to North America. Here again, upon detection the populations were treated swiftly with herbicide, the site was searched annually for residual plants until 1992 (corres. of B. Whyte, Queensland Department of Lands), and specimens were routinely collected (T. A. Cole (s.n.), Alan Fletcher Research Station, Brisbane, Queensland). Both these Queensland examples of eradication are remarkable because the non-indigenous species were unknown in Australia beforehand, and the species have not been detected in the country since eradication campaigns were launched against them.

Action on a similar scale led to the eradication of the perennial herb, *Centaurea trichocephala*, in the western United States. At least eight *Centaurea* spp. are already naturalised, if not invasive, in the U. S. (Whitson *et al*. 1996). As a group, they are considered among the most noxious weeds in the arid western U.S. because they readily invade rangelands, provide no forage for livestock, and their control is difficult, if not intractable (Watson and Renney 1974). The reputation of the congeners led to unusually swift action when a previously-undetected *Centaurea* species was identified near Tampico (Yakima County), Washington in 1985. Once identified, the only known population (approx. 300-400 adult plants) was destroyed with herbicide in 1986. Additional herbicide treatment followed in 1987, and the site was inspected annually for any remaining *C. trichocephala* through 1990. In 1990 the Yakima County Noxious Weed Board made an official declaration that *C. trichocephala* had been eradicated from its only known site (M. Slaugh, pers. comm.). No other plants have been detected in the United States.

Salvinia molesta, the highly-aggressive tropical aquatic invader (Thomas and Room 1986), was not discovered in the United States (apart from in a few botanical gardens) until 1995 (Myers 1982 as cited in Nelson 1984). Its first field detection was in a 0.6 ha pond near Walterboro (Colleton County), South Carolina (Johnson 1995); conditions from which it could have spread readily. Eradication began several months after its detection. Hand removal of *S. molesta* in the pond was followed by herbicide application by the South Carolina Department of Natural Resources and the federal Animal Plant Health and Inspection Service (APHIS). As a result, the species was eradicated from the site and potentially from the U.S. (R. G. Westbrooks, pers. comm.). Unfortunately, this victory was short-lived: *S. molesta* appeared in Texas in 1997 and in more states in 1999. Most recently it has been found in North Carolina, plus new locales in other states, including Hawaii (Anon. 2000, http://nas.er.usgs.gov/plants/sa_molesta/docs/sa_mol.html). Its further spread in the U.S. is certain without extraordinary effort at containment and eradication, which may already be unattainable.

New Zealand has been exemplary in its successful national eradication programmes for non-indigenous terrestrial and aquatic species. So far, the terrestrial species *Acroptilon repens* and *Chondrilla juncea*, and the aquatic macrophytes, *Zizania palustris*, *Menyanthes trifoliata*, and *Pistia stratiodes*, have been eradicated from all known sites in the country (P. D. Champion, pers. comm.). Furthermore, the scourges *Sorghum halepense*, *Eichhornia crassipes* and *Salvinia molesta* have been reduced, such that the only occurrences are newly-detected sites. For *E. crassipes* and *S. molesta* the new sites are presumably the products of their illegal culture. All these eradication efforts are testimony to the zeal that New Zealand has applied in scrutinising the entry of non-indigenous species, then controlling and eliminating those that prove harmful under its biosecurity legislation, as consolidated in the Biosecurity Act of 1993 and its amendments (http://rangi.knowledge-basket.co.nz/gpacts/public/text/1993/an/095.html).

Other eradication programmes are underway; for some, the objective appears to be attainable. It is remarkable that the once deliberate spread of the salt-tolerant shrub *Bassia* (or *Kochia*) *scoparia* in Western Australia has not only been halted but even reversed. Although this Eurasian species was well known as an aggressive weed in North America (Whitson *et al.* 1996), dubious claims as to its forage value (e.g., Grimson *et al.* 1989; Mir *et al.* 1991) led to its introduction in Western Australia in 1990. Soon sufficient grower concern had arisen about its invasive character that its sale was halted, and control measures were begun in 1992. From an estimated maximum extent of 3277 ha in 1993, the new Australian range of *B. scoparia* has been reduced to about 5 ha. The plant has not reappeared in treated areas for at least three years (Wittenberg and Cock 2001). As laudatory as this effort has been, a stringent standard for eradication (total destruction of all detectable plants), is necessary. Too often, a non-indigenous species has been on the verge of eradication, the effort has slackened, and the species has rebounded in its new range (e.g., *Berberis vulgaris* in the U.S., Mack 2000). As satisfying as the reduction of an invasion to a few small remnant populations is, these remnants may become the nascent foci from which a re-invasion can emerge (Moody and Mack 1988; Higgins *et al.* 2000).

Even though these examples are drawn from widely-separated areas and involve taxonomically-unrelated species, these eradication campaigns share common traits. The non-indigenous species consisted of one, or at most a few, small populations. Eradication efforts began promptly after first detection, sometimes (as in the case of *S. molesta* in South Carolina) a few months later. And equally important, there were repeated surveys of the treated site(s) to detect and destroy any new or previously overlooked plants. Eradication, not simply control, was recognised as attainable, provided the effort was initiated quickly. These examples include both island and continental case histories: islands may offer some advantages in eradication, given their geographic and ecological boundaries, but even a non-indigenous species on a continent can be eradicated if action is swift.

STRATEGY COMPARED WITH TACTICS: ATTENTION TO NASCENT FOCI

The spatial pattern of adventitious and naturalised species falls into two functional categories: species that reproduce but do not expand their range beyond the point of introduction (e.g., *Rotala indica* at the Biggs Experiment Station, Davis, California (USA), Barrett & Seaman 1980), and those that disperse into new locales. A non-indigenous species with many widely-separated foci is much more difficult to eradicate than a single infestation. But detection and eradication of all nascent foci of a potential invasion are more important than attacking large centres of the infestation, whether control or eradication is the goal. Ignoring these small foci of an introduced species, while treating only major infestations in a new range, simply provides time for these once-inconspicuous populations to flourish. This contention, which seems at first counter-intuitive, has been amply illustrated in theory (Moody and Mack 1988; Higgins *et al.* 2000) and more importantly demonstrated in successful control campaigns that may yet be transformed into eradication.

The pending eradication in the United States of the introduced herbaceous hemiparasite, *Striga asiatica*, through emphasis on the destruction of nascent foci, is remarkable. The prospects for control, much less containment and potential eradication, would have seemed bleak when this aggressive plant parasite was first detected in North Carolina in 1956. The plant had already infested maize in a four-county area and would subsequently be detected in northern South Carolina (Westbrooks and Eplee 1999). In addition to establishing an effective quarantine to the export of *S. asiatica* seeds from the infected region, APHIS destroyed outlier populations and nascent foci before attempting to shrink the main infestation (Eplee 1981). Through this strategy, the invasion had been reduced in 1999 to about 6000 ha from its maximum extent of 177,000 ha in the early 1960s (Westbrooks and Eplee 1999). Across four decades the cost of destroying *S. asiatica* in the U.S. has totalled USD250 million (R. E. Eplee, pers. comm.): good value compared with the crop losses an unchecked *Striga* invasion would have caused. Much greater savings would have been realised over the last 50 years, however, had the same money been applied to a national early detection/eradication network that would have included the detection and eradication of all damaging non-indigenous species, including *Striga,* upon their entry.

Eradication may not be attainable in other cases, but attacking nascent foci nevertheless remains the key to the areal containment of the invader. *Casuarina equisetifolia* and *Melaleuca quinquinervia* are devastating tree invaders within the Everglades, an ecological and floristic island in southern Florida. They each occupy 100,000 ha of habitat (Schmitz *et al*. 1997). Despite the immense infestations that each species forms, emphasis in their effective control has centred on eradicating their nascent foci (Doren and Jones 1997). Slowly, these invasions may be first blunted and then potentially shrunk; attention can be then shifted to destruction of centres of the infestation.

Perhaps more challenging has been the attempt to control spread of *Mimosa pigra* (catclaw mimosa) in the Northern Territory, Australia. Its dispersal has been aided by water transport of its seeds, and until recently, by the introduced Asian water buffalo, which is itself the subject of a control/eradication programme (Lonsdale *et al.* 1989). The ability to detect nascent foci in the Australian new range of *M. pigra* is even more daunting than for introduced trees in the Everglades – the potential area of search is enormous and remote. Nevertheless, multi-year control of *M. pigra* in these satellite populations has successfully prevented the development of large stands of the invader in Kakadu National Park (Cook *et al.* 1996).

Attention to small foci has also contributed to containing catclaw mimosa in southern Florida. The potential for its invasion remains high; it currently occurs in approximately 395 ha (Schmitz and Westbrooks 1997), scattered across three counties. So far, only two populations have been eradicated, but almost 20,000 plants are destroyed annually. Eradication of one of the populations involved removing all seeds from the 20-30 trees in the stand, felling the trees, treating the stumps with herbicide, and surveying the site for post-treatment seedlings (R. Kipker, pers. comm.). Eradication is an attainable goal for *M. pigra* in southern Florida, but 15 years into the campaign, it is clear this will be a long-term venture. Other long-term eradication campaigns include *Chromolaena odorata* (eight years and continuing) and *Orobanche ramosa* (five years and recommended to continue for at least 10 more years), both in Australia.

The detection and destruction of outlier, and particularly nascent isolated populations, remain among the most important lessons learned from attempts to eradicate potentially invasive species. This lesson applies equally well to invasions anywhere – islands or continents.

POST-ERADICATION SURVEYING AND RE-ERADICATION: THE ESSENTIAL POSTSCRIPT

In a strict sense, use of the term 'eradication' in combating non-indigenous plants can be a misnomer. It can be confidently applied to those cases in which all individuals of a species are destroyed in a new range (Westbrooks 1993), but more commonly it refers to destruction of individuals below a level of detection. This distinction is important because even if all vegetative plants are destroyed, the species may persist in a seed bank. Repeated surveying of the area is essential, no matter its size, to detect remnant or newly-emergent individuals.

Post-eradication surveys for non-indigenous species have a distinguished but apparently long-forgotten past. The 1876 centenary celebration of the United States included an international exhibition in Philadelphia at which many nations exhibited their country's livestock and crops. The United States Centennial Commission had the remarkable foresight to recognise this venue as an opportunity for the inadvertent introduction of non-indigenous species. Local botanists carefully surveyed the exhibition grounds for *four years* after its close, in order to detect any immigrant species that might gain a foothold. Thirteen adventives were detected (and presumably destroyed); these included the herbs *Crepis tectorum*, *Centaurea nigra* and *Lepidium sativum* (LeConte *et al.* 1881). Had this episode sparked diligent early detection and eradication of inadvertent plant immigrants in the U.S., the scope and magnitude of invasions into the U.S. in the following 120 years might have been much different.

As noted above, repeated survey of the site of a putative eradication is not only prudent but should be mandatory. The initial eradication effort for *Eupatorium serotinum* and *Helenium amarum* in Queensland did not completely eliminate these immigrant species. Had the detection sites not been repeatedly inspected and treated, these species might still be in the Queensland flora and even naturalised. Similar attention to survivors has been a hallmark of successful eradication efforts.

While repeated surveying becomes much more difficult as the treated area becomes larger, it is still obligatory. In the ambitious programme to restore large fractions of the Cape Floristic Province in South Africa to their native floristic condition, all invasive species are progressively removed or destroyed *in situ* within much of the Cape Peninsula National Park. Since the early 1980s, each block in the extensive eradication design has been progressively cleared of invasive plants, then surveyed thoroughly every two years to destroy any plants emerging from these species' depleting seed banks. By the mid-1980s, more than 6700 ha had been treated in this manner (Clark 1985, Macdonald 1989). This re-surveying has continued. One of us (RNM) viewed restored sites in 2000; no invasive species were detected.

The distinction between eradication and control, albeit diligent control, blurs where there is a continual threat of the re-entry of invasive species. The vegetation of Bermuda has been almost totally transformed by introduced species (Wingate 1992). Particularly devastating was the 1940s entry of North American scale insects, which rapidly reduced the once-prominent Bermuda cedar *Juniperus bermudiana* to a few remnant trees (Challinor and Wingate 1971). Unfortunately, re-establishment of forest cover in Bermuda followed the same foolhardy practices adopted elsewhere (Mack 2001); native species were ignored in the name of expediency to re-establish forest cover. As a result, subtropical trees were introduced, principally *Casuarina equisetifolia* (Nolan 1980), which has in the past decade become invasive (D. B. Wingate, pers. comm.). Worse still, *Schinus terebinthifolius* (Brazilian pepper) escaped from gardens in the 1950s and became invasive (Challinor and Wingate 1971). Today, *S. terebinthifolius* is invading the upland margin of Bermuda's mangrove swamps (Thomas 1993), as well as other habitats. Introduced starlings (*Sturnus vulgaris*) repeatedly carry the bright red fruits of *S. terebinthifolius* throughout the island group. As a result, even tiny islands offshore can be readily re-infested with *S. terebinthifolius* (D. B. Wingate, pers. comm.).

Nonsuch Island is only 6 ha, yet it is important as a reserve of natural vegetation (Wingate 1992). The Bermuda Department of Agriculture has removed all *S. terebinthifolius* (along with *Asparagus densiflorus*, *Livistonia chinensis*, *Pimeta dioica*, *Eugenia uniflora* and *Citharexylum spinosum*) on Nonsuch Is. for more than 20 years. Eradication has been possible for all species, except the bird-dispersed *S. terebinthifolius* and *A. densiflorus*. Seedlings of all introduced species are removed as they appear. Any delay in removal only diminishes the likelihood of eradication; for example, two-year old Brazilian pepper will produce seeds. As many as 500,000 Brazilian pepper seedlings must be removed on this small island annually in order to hold the invasion in check (D. B. Wingate, pers. comm.).

IS SINGLE SPECIES ERADICATION WORTHWHILE? THE SISYPHUS EFFECT

In addition to the trials and tribulations of ever achieving a species' eradication, removal of a single species can have an unintended effect. In conservation biology, the implicit goal in control or eradication of invaders is the prospering of native species (Van *et al.*1998). But does that outcome always occur? Could the demise of the target species instead facilitate the emergence in the community of another long-suppressed non-indigenous species? We liken this hypothetical phenomenon to a Sisyphean experience: success (eradication) brings failure (another non-indigenous species is inadvertently fostered). So far, this general hypothesis has not been dissected into testable hypotheses, much less experimentally examined. We do have, however, observations from widely-scattered locales that could be explained by a scenario in which a non-indigenous species became more abundant coincident with the control or eradication of another non-indigenous species:

■ The coincident increase of one non-indigenous species with the decline in another was quantified during the biological control of *Hypericum perforatum* (St. John's wort) in California more than 40 years ago. As *H. perforatum* steadily declined in plots through the 1950s through attack by *Chrysolina quadrigemina*, other non-indigenous species became more abundant, including the now abundant invader *Centaurea solstitialis* (Huffaker and Kennett 1959). Tisdale (1976) maintains that a similar emergence of invasive annual grasses (primarily *Bromus tectorum* and *Taeniatherum caput-medusae*) occurred on sites in northern Idaho once dominated by *H. perforatum* as *C. quadrigemina* reduced the biomass of St. John's wort from 1110 kg/ha to less than 100 kg/ha.

■ *Acacia saligna* is being controlled in South Africa with the release of the fungus *Uromycladium tepperianum* (Morris 1999). Although *A. saligna* and *Acacia pycnantha* are not sympatric in their native ranges in Australia (Costermans 1983), they share a common ecological amplitude. And the two wattles can occur in the same habitats in South Africa (J. H. Hoffman, pers. comm.). As a result, *A. pycnantha* could occupy sites once dominated by *A. saligna*. Fortunately, *A. pycnantha* is itself controlled by a hymenopteran, *Trichilogaster* sp. (Dennill et al. 1999; J. H. Hoffman, pers. comm.) so this replacement may be fortuitously blunted.

■ Two of the most destructive invaders in the tropics and subtropics, *Chromolaena odorata* and *Lantana camara* also commonly occupy similar habitats throughout their huge new range, including sites in India (Muniappan *et al.* 1989) and southern Africa (cf. distribution maps in Baars and Neser 1999; Zachariades *et al.* 1999). Consequently, the removal of one through any form of control provides opportunity for increase in cover of the other (J. H. Hoffman, pers. comm.).

■ In South Africa the invasive leguminous tree *Sesbania punicea* has been brought under effective control with release of three phytophagous insects (Hoffman and Moran 1999). Unfortunately, in several of the sites monitored in this control effort, decline of *S. punicea* has coincided with rise in the abundance of *Lantana camara* (J. H. Hoffman, pers. comm.).

Similar events occur amongst invasive aquatic macrophytes. These species often invade communities that have few if any native macrophytes; for example, there is a conspicuous paucity of floating macrophytes in the Australian flora (Jacobs 1999). The low species richness of vascular plants in these communities lends itself to detection of changes in the abundance of a handful of non-indigenous species (Mack 2000). As is apparent with the terrestrial examples cited above, the results suggest (but do not demonstrate) that the removal of one invasive species sparked the rise in abundance of another.

■ Lake Seminole is a large impoundment created in 1957 by damming the Chattahoochee and Flint Rivers in Florida. Soon after the lake's creation, *Eichhornia crassipes* entered, and by 1960, covered approximately 2500 ha of the now-submerged Flint River arm of the lake. Extensive herbicide treatment from 1960 through 1962 devastated water hyacinth; coincident with its decline, another invasive macrophyte, *Alternanthera philoxeroides* soon expanded its coverage. Alligator weed was attacked subsequently by a flea beetle (*Agasicles* sp.) in 1968, and its coverage declined. Apparently as a result, water hyacinth rapidly re-expanded its coverage from a few hectares to 2030 ha in less than a year. By 1975 both these invasive species were still in the Flint River arm of the lake, although in reduced coverage. Simultaneously, the native grass, *Zizaniopsis miliacea*, was becoming prominent along the shoreline (Anon. 1980). Through a combination of herbicide application and insect grazing, the populations of these two invasive aquatic macrophytes have radically fluctuated for over 20 years, each taking up the slack left by the other.

■ Elsewhere in southern Florida, the widespread invader *Hydrilla verticillata* has been controlled in canals with a combination of grass carp and herbicides (R. Stocker, pers. comm.). But its reduction in coverage has coincided with an increase in the equally-unwanted invader *Hygrophila polysperma* (Sutton 1995).

■ The ability of one invasive aquatic macrophyte to increase its role as another is brought under control has long been recognised in the biological control of these species in Australia. Consequently, programmes were undertaken simultaneously for the biological control of the four most prevalent aquatic invaders (*Eichhornia crassipes*, *Pistia stratiodes*, *Salvinia molesta*, and *Hydrilla verticillata*) (Harley 1988).

Even though most of the examples involve species that were the target of a biological control programme, biological control is not in any sense a target of our query here. Similar results arise with selective herbicide treatment, especially in crop fields. Application of broad-leaf herbicides would readily cause competitive release among grasses and other weedy monocots (Aldrich and Kremer 1997). Such an outcome may also be rendered less likely by simultaneous application of multiple or broad-spectrum herbicides that remove potential replacement species among the non-indigenous flora.

CONCLUSIONS

Eradication is almost totally dependent on the early detection of a non-indigenous plant species in a new range, including an island.

In setting priorities within an eradication effort, emphasis should first be placed on destroying the immigrant's nascent foci, rather than the oldest or most-concentrated population(s) in the new range.

Even after the species has putatively been eradicated, continual surveying for it is essential. A lapse in such surveying could allow remnant foci to grow and undo the entire eradication effort. Eradication campaigns for invasive plant species typically need to last for 10 years or more.

Because the prediction of a species' invasive ability is problematic, it is better to err on the side of eradicating a non-indigenous species that later proves to be innocuous, than to withhold eradication until the species' fate is clear (Lonsdale & Smith 2000). Delay greatly reduces the prospects for eradication.

If the goal in eradication is environmental conservation, attention should focus on creating a *zone sanitaire*, rather than on undertaking single species removals, and on establishing the desired native vegetation as rapidly as possible.

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Eradication of Pacific rats (*Rattus exulans*) from Whenua Hou Nature Reserve (Codfish Island), Putauhinu and Rarotoka Islands, New Zealand.

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Abstract Pacific rats (*Rattus exulans*) were eradicated from three islands in southern New Zealand in 1997 and 1998. In August 1998, two aerial applications (9.7 kg/ha and 9.4 kg/ha) of 2 g Agtech cereal pellets containing 20 ppm brodifacoum, were made over the whole of 1396 ha Codfish Island (Whenua Hou Nature Reserve) for the eradication of Pacific rats *(Rattus exulans)*. Preparation for the eradication included the eradication of Pacific rats from 144 ha Putauhinu in August 1997 so that a second population of a fernbird (*Bowdleria punctata wilsoni*) endemic to Codfish Island could be established there. A bait drop was also carried out on 88 ha Rarotoka (Centre Island) on the same day as Putauhinu, to eradicate Pacific rats as the first step in the restoration of that island. A single drop of 12 kg/ha of Talon 20P cereal bait containing brodifacoum at 20 ppm was used for the Putauhinu and Rarotoka eradications. To protect the fernbirds on Codfish Island which were at risk from the aerial bait drop, a ground bait station network on a 50 m x 25 m grid covering 37 ha was set up and operated in conjunction with the aerial drop. With no sign of rats having been detected for two years after the respective bait drops, the three eradications have been declared successful.

Keywords Pacific rat, kiore, *Rattus exulans*; eradication; brodifacoum; fernbird, *Bowdleria punctata*.

INTRODUCTION

Whenua Hou Nature Reserve (1396 ha), also known as Codfish Island, is located three kilometres west of Northern Stewart Island (Fig. 1). The New Zealand Department of Conservation (DOC) identified Codfish Island as a high priority site for the eradication of Pacific rat or kiore (*Rattus exulans*) prior to 1995, when planning for the eradication started. DOC managers had requested the eradication of Pacific rat as a further step in the ecological restoration of the island. Codfish Island is the largest island near the South Island and Stewart Island, which had such easily-realised restoration potential. The island has a diverse range of habitats typical of the Stewart Island area (Rakiura Ecological Region).

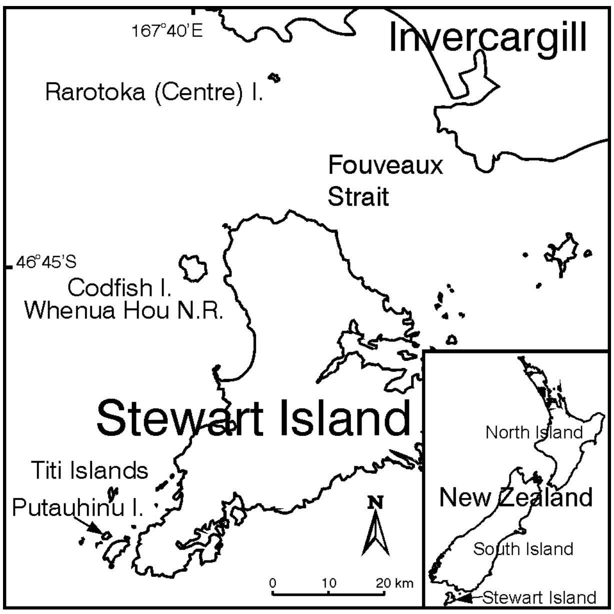

Fig. 1 The locations of Rarotoka, Putauhinu and Codfish islands.

Pacific rats are believed to have been introduced to Codfish Island by early Maori, possibly for use as a food source during their annual harvesting trips to the nearby Titi Islands. Like most New Zealand Maori, the local people consider Pacific rat a ‘taonga’ or treasure as they were introduced to New Zealand by the first colonists. However, muttonbirders, those Maori with inherited rights to harvest muttonbirds (young of sooty shearwaters (*Puffinus griseus*)), generally consider the Pacific rat as a pest to be removed, providing the species is safeguarded on some islands. For most local Maori the presence of Pacific rats on Stewart Island fills this requirement.

A range of trials was required prior to the rat eradication being undertaken on Codfish Island. Trials included: bait and toxin weathering; identification of non-target species at significant risk; and identification of the appropriate management techniques to minimise this risk. As a result of these trials, DOC determined that Pacific rat eradication was required on Putauhinu, a 144 ha island south-west of Stewart Island. This eradication was to prepare Putauhinu for the establishment of a second population of one of the non-target species at risk, the Codfish Island fernbird (*Bowdleria punctata wilsoni*).

Putauhinu is privately owned by Maori and is visited annually in autumn for up to two months by five families to harvest muttonbirds. For its size, Putauhinu contains a diverse range of habitats from tall rata (M*etrosideros*) forest to low flax (*Phormium* spp.) pakahi (wetland), but before the eradication, it had a very depleted fauna due to the presence of Pacific rats and cats (*Felis catus*). The cats

Pages 173-181 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

died out in the 1960s-1970s after some limited control by the muttonbirders (T. Davis pers. comm.).

The muttonbirders on Putauhinu first approached the DOC about removing Pacific rats from their island in 1994, in order to continue the restoration of the island to its former condition. Therefore, they were happy with DOC's proposal for the rat eradication and introduction of fernbirds to Putauhinu. Fernbirds had previously been on the island but had been exterminated by the rats and cats.

During the initial planning stages for the Putauhinu eradication the local Maori also suggested the possibility of eradicating Pacific rats from Rarotoka (Centre Island), an 88 ha island in Fouveaux Strait (Fig. 1). Rarotoka is a former lighthouse reserve which was cleared and intensively grazed for more than 100 years by sheep (*Ovis aries*) and cattle (*Bos taurus*) prior to being handed back to its original Maori owners. The rat eradication on Rarotoka was proposed as the first stage in the ongoing restoration of the island, which will include intensive plantings of native species.

METHODS

Putauhinu

The eradication operation on Putauhinu was carried out in August 1997 using a single aerial bait drop of 12kg/ha of Talon 20P bait (containing 20 ppm brodifacoum). The bait was loaded into the helicopter's spreader bucket directly off the deck of the boat which had been used to transport the bait to the island on the day of the drop. A second helicopter was used to transport fuel and personnel.

As the island has five dwellings which are occupied seasonally and rely on rainwater, a team was tasked with placing bait under and in all buildings and with disconnecting the water pipes to eliminate the risk of contamination.

Rarotoka

The eradication operation on Rarotoka was also carried out using a single 12kg/ha aerial drop of Talon 20P. Proximity to the mainland meant that it was more efficient to ferry the bait directly from there using a second helicopter, than to use a boat. Bait loading was carried out on the island with a single helicopter dropping the bait.

The relatively small size of Putauhinu and Rarotoka, and their proximity to each other, meant that eradication operations on them could be done cost effectively in the same day using the same helicopters and personnel.

As the risk that bitrex posed to the success of these operations (Veitch 2002) was not recognised at this stage the bait used for this drop did contain bitrex.

Codfish Island

Preparation for the eradication operation on Codfish Island began in 1992 with the commencement of a number of trials to gauge the likely effects of the operation on non-target species and to find the most suitable bait type.

Assessment of risks to non-target species

Fernbirds

Fernbirds had not previously been identified as a non-target species that would be affected by an aerial bait drop. However, this was based on toxins other than brodifacoum. As Codfish Island fernbirds are recognised as a separate subspecies, DOC staff decided it was necessary to quantify any possible risk. To this end the relevant permissions were obtained for a toxic trial at a mainland site. In 1993 toxic bait was dropped over 25 ha and the fernbirds monitored. Over 80% of the fernbirds in the area disappeared and one fledgling, which was found dead, was analysed and found to contain brodifacoum (Ranum *et al.* 1994). No birds in a neighbouring control block were lost, thereby showing that the proposed aerial bait drop on Codfish posed a significant risk to the fernbirds at a population level. In 1996 a repeat trial was carried out using bait stations (Russell and Parker 1997). The bait station trial had no observable effect on the fernbirds.

Also in 1996 a trial was carried out on the mainland to see if it was feasible to hold sufficient fernbirds to safeguard the subspecies in captivity for long enough to ensure that when they were released, they would no longer be at risk from residual baits. While birds could be successfully held in captivity, the territorial nature of the birds meant that it was not feasible to hold the required numbers as subordinate birds died apparently of stress-related causes.

Kakapo

Field trials using non-toxic bait were carried out on critically-endangered kakapo (*Strigops habroptilus*) which indicated that some birds might be at risk from eating the bait. At least one bird ate non-toxic bait when presented with it.

A non-toxic aerial bait drop on Codfish Island to determine risk to bats and kakapo was unsuccessful as the bait was washed out on the night of the drop by a heavy rain event.

Bats

To gauge the likelihood of secondary poisoning of short-tailed bats (*Mystacina tuberculatus tuberculatus*), brodifacoum cereal baits were fed to weta (*Hemiandrus* spp.), a large (20–40 mm) orthopteran that is part of the bats' natural diet. The weta were then assayed for the toxin. The assay showed that brodifacoum passed through a weta digestive tract in less than twelve days (Lloyd 1997). This meant that the likelihood of secondary poisoning to bats was minimal, as a bat would need to eat a large number of invertebrates which had in turn each eaten a large amount of bait in the recent past.

Bait was presented directly to captive short-tailed bats but none was eaten (Lloyd 1997). However, as aerial bait drops had not been carried out on an island with bats before, DOC decided to hold a backup population in captivity and to monitor the wild population.

Fifty bats were transferred to nearby Ulva Island (40 km away) in the hope of establishing a backup population (Lloyd 1997). However, no bats were recorded after the "hard" release (i.e. the bats were released immediately on arrival at Ulva with no shelter or food provided) and much more work would have been required to make this technique work.

Trials were carried out to see if bats could be held in induced torpor, in a fridge, to reduce the feeding requirements during the period in captivity (Lloyd 1997). While the trial was successful it was not judged a practical option for the number of bats required (i.e. 300-400) to safeguard the population.

Other Species

The effect of an aerial brodifacoum drop on other non-target species had been noted during previous operations (e.g. kaka (*Nestor meridionalis*) on Kapiti, Empson and Miskelly 1999), and while some deaths were anticipated, any effect was expected to be on an individual (particularly juvenile birds), rather than population, level.

Bait Trials

DOC staff were concerned about the possible effect on non-target species of having bait available on the ground for a prolonged period; however to increase the chances of the eradication succeeding, the bait was required to be available to the rats for as long as possible. Therefore, trials were carried out to find the best compromise between the two conflicting issues.

Bait weathering

A comparison between Wanganui No. 7 and Agtech baits to test longevity of both baits under the climatic conditions found on Codfish Island was carried out in 1994. The longevity of Agtech bait was less than Wanganui No. 7 bait, but it could still handle at least 15 mm of rain. The longevity of the bait is, initially at least, based solely on precipitation levels; not on how long the bait is on the ground. Agtech was the preferred bait because, with reasonable weather, it would last long enough for all rats to have access to bait (minimum 3 nights), but would be unlikely to last for an extended period (i.e. over a month). The faster breakdown of Agtech bait reduced the length of time that bait would be available to the high-risk non-target species such as bats and fernbirds. The faster breakdown also reduced the risk to more common non-target species such as kaka, which Empson and Miskelly (1999) discovered could "learn" that the bait is food, the longer it is available. Faster bait breakdown also meant that kakapo and captive bats could be released on the island sooner, reducing both the stress on the individuals and the cost of holding them.

Toxin weathering

To find out at what point the bait was no longer likely to be lethal to rats and non-target species, bait was put out on Codfish Island under rat exclosures and samples analysed as it broke down. This showed that even when the bait appeared unpalatable (i.e. either mouldy or crumbled), it still contained the original toxic loading.

Selection of a quicker breakdown bait meant that the safety margin usually allowed for weather conditions in such an operation was reduced. This reduced safety margin meant that more emphasis was placed on accurate weather forecasting as >15mm of rain within three nights of the bait drop could have resulted in a failure of the entire operation.

Timing of operations

In New Zealand, rat eradications are undertaken in the winter when rat numbers are at their lowest (shown for Codfish by a three year, monthly trapping programme (DOC internal file REN 012)) and natural food is presumed to be in shortest supply. Also, in the winter rats are not breeding, therefore minimising the risk of any young rats being in the nest when bait is dropped and emerging once the bait has broken down.

A weather forecast for three nights without rain is also considered a requirement for all aerial bait drops. However, three consecutive fine nights is not always easy to obtain in southern New Zealand, particularly during an unsettled winter such as 1998.

Bait requirements

Well into the preparation for the Codfish Island operation DOC became aware that the toxin (brodifacoum) which was to be used contained bitrex, which has been incorporated into all Talon bait formulations (including the Agtech bait proposed for use in this operation), since late 1996. Bitrex was added to make the bait less palatable to humans, and the bait registrations had been altered accordingly, meaning that the selected bait could not be legally used without having bitrex in it. This additive had been put in the bait for previous operations without DOC's knowledge or approval despite requests being made for notification of any changes to formulation. While company trials had shown that bitrex may in fact act as an attractant to possums and some rodents (D. MacGibbon, ICI, pers. comm.), two out of four of the previous eradication attempts using bitrex had failed (I. McFadden pers. comm.), and the outcomes of the other two operations were unconfirmed at the time. In laboratory efficacy tests, involving bitrex in ICI rodenticidal formulations with albino rats and mice (20 animal groups, three-day choice tests), three out of 60 rats rats did not eat sufficient bait to be killed (Kaukeinen and Buckle 1992). In tests on Pacific rats on Little Barrier Island, 12 of the 15 rats offered a choice of baits with (and without) bitrex, chose to eat significantly less of the bait containing bitrex (Veitch 2002). This was sufficient for DOC to stipulate that bitrex could not be present in the bait for this operation.

While most of the other specifications for the bait are standard (i.e. colour, moisture content, etc.). bait size was an important consideration. Twelve mm diameter baits (approximately 2 g), were selected as this gave the greatest number of individual baits per square metre, increasing the chance of each rat encountering bait.

Describing bait by weight/mass is in fact not appropriate, as bait production standards are given using diameter; in this case 12 mm. Depending upon factors on the day of manufacture such as humidity, etc., there can be a significant variance in the weight of each bait. Hence the cited weight of 2 g per bait should only be taken as a rough guide. In New Zealand, bait is available in 12, 16 and 20 mm diameters.

Bait storage

DOC staff decided that the bait should be stored on site until it was required, therefore temporary storage was erected using a pipe framed "Coverall" tent. This proved more than adequate but did require daily monitoring to ensure adequate ventilation and to avoid condensation falling onto the bait.

Helicopter operations

Due to uncertainty that a single helicopter could complete the drop in the time required (i.e. one day), two helicopters were arranged for the first drop along with a backup approximately one hour flying time away on the mainland. The island's boundary (i.e. the shoreline) and the core area (containing the fernbirds) that was to be treated using ground bait stations, were logged in with the GPS the day before the drop. These data were then stored in the helicopters' computers so that boundary logging did not need to be repeated for the second drop.

For safety, both to reduce the risk of collision and to speed up the operation (i.e. to reduce ferrying time from the loading site to the drop areas), the first drop was carried out using two loading sites. One site was at the bait storage area on the coast, and a second near the summit of the island. Bait was ferried to the top site by a third helicopter, which also transported personnel and the media representatives as required. The first drop showed that the island could be covered in one day by one helicopter, therefore the second drop was carried out by a single bait-drop helicopter operating from the top loading site, with a second helicopter ferrying bait to that site.

Each loading site had a platform made out of wooden pallets on which 12 opened bags of bait were placed so that loading could take place as quickly as possible. Having a raised platform allowed the bait loaders to tip the bait in quickly, rather than lifting each bag from ground level.

To monitor the bait coverage a Trimble GPS system, which recorded the lines that the helicopter had flown, was used. However, the pilot had to manually switch the tracking on while bait was flowing out of the bucket and switch the GPS off when bait flow stopped.

Bait drop

The Codfish Island bait drop was carried out to the formula (two bait drops, with a 20% overlap in flight lines for each drop) that had been used successfully elsewhere (I. McFadden pers. comm). The cliff areas (Fig. 2) were treated twice during each drop to allow for the increased planar area that had to be covered. This system ensured full coverage of the island and sufficient bait on the ground to ensure that every rat had access to a lethal dose.

The first drop was to be 8 kg/ha, followed by a second of 4 kg/ha. The first drop on 18 August 1998 ended up averaging 9.7 kg/ha due to the double-up on the cliffs, around the bait station area, and treating the rock stacks and small islets offshore, as the presence of Pacific rat on these could not be ruled out.

Because it rained the night of the first drop (5 mm) and again on the second day (7.1 mm), the DOC project team on the island decided to increase the second drop to 8 kg/ha due to concern about the possibility of the baits weathering. The second drop on 27 August 1998 averaged 9.3 kg/ha. The increase over the 8 kg/ha originally proposed was to ensure complete cliff coverage.

Bait was spread over the entire island, excluding the core fernbird area. This included the buildings, which had the water systems disconnected and were later washed down, and the bat aviaries, which were covered with polythene during the drop. No bait was dropped on the core fernbird area, but the perimeter of this area was flown with a double swath to ensure that the border between the two techniques was secure. The steep/cliff areas of the island were covered twice to ensure that they received the appropriate drop rate. Bait was also spread over all the islets and rock stacks adjacent to Codfish Island.

Accuracy of aerial bait spread

The basic requirement for the operation to succeed was that every rat had access to a lethal dose of bait (i.e. that bait was dropped into every rat's home range). To ensure that the bait drop covered the whole island a Trimble® second-generation differential global positioning system was used by the pilots to record the flight path of the helicopters. This allowed highly accurate bait placement on all parts of the island, including the steep coastal faces and adjacent rock stacks and islets. A map of the bait spread was available on a computer screen in the helicopter that could be viewed when required. This took place at least every time the helicopters refuelled. The information was also downloaded after the drop and presented as a printed map. The map enabled the project manager to locate any possible gaps in the bait spread and get the pilots to cover those areas again. A print out of satellite flight paths was obtained prior to the drop to ensure that there was suitable GPS coverage.

Safeguarding kakapo

As a precautionary approach, all known kakapo on Codfish Island were removed to a safe island for the duration of the operation. One male who could not be found at the time of the transfer was located later, having been on Codfish Island for the duration of the operation, with no apparent ill affects.

Safeguarding fernbirds

Early trials confirmed that the endemic Codfish Island fernbird was likely to be seriously affected by an aerial drop of cereal bait, and that a ground bait station operation presented far less risk. A ground-based operation over the whole island was not deemed feasible as it would have had significant impacts on the wildlife and their habitats, including the cutting of many kilometres of track over fragile country. Further, using more ground bait stations would not guarantee exposing all rats to a lethal dose of bait.

It was therefore decided to use a two-pronged approach to protect the fernbirds:

Bait station network

A 37 ha block of the best fernbird habitat (Fig. 2), containing the densest population of fernbirds determined by a previous survey (G. Elliott pers. comm.), was set out with a total of 416 bait stations at 25 m intervals along parallel lines 50 m apart. Around the perimeter of the grid, the intervals between bait stations was reduced to 25 m x 25 m. Each bait station was loaded with 100 g of Agtech pellets on the day prior to the first drop to ensure that more than sufficient bait was available to the rats. After six days this was reduced to 10 pellets to enable easier monitoring of any bait take. After a further 31 days this was changed to a 28 g Contrac wax block containing 50 ppm bromadiolone in case of bait or toxin shyness (Table 1).

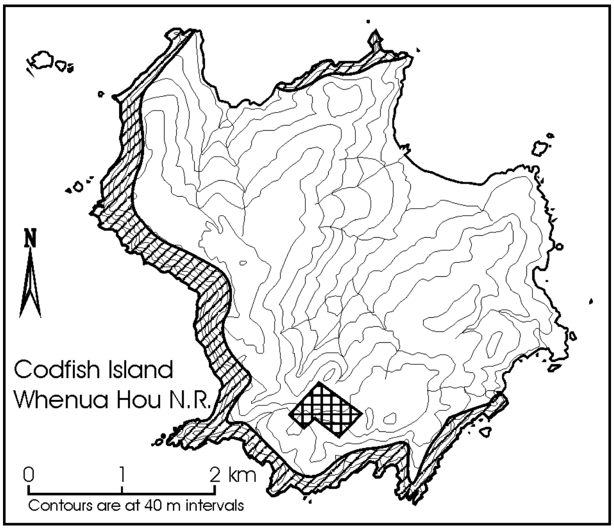

Fig. 2 Codfish Island showing the cliff areas (diagonal hatching) and core fernbird area (cross hatching) which were given specific treatments in the rat eradication programme.

The bait stations were made of 100 mm diameter Novacoil plastic drainage pipe, 450 mm long with a hatch in the middle for loading and checking bait. To help the helicopter pilots identify the boundary of the zone during the aerial application of bait, the perimeter was flagged using brightly coloured pennants at approximately 50 m intervals.

During both aerial applications the core area was excluded from any aerial poisoning. However in addition to the normal drop up to the boundary, two 60 m swaths were dropped around the entire boundary on both drops to reduce the possibility of rats moving in or out of the ground treatment area. The inner swath was centred on the perimeter flags, which meant that this bait dropped from the air penetrated 25–30 m into the grid from all boundaries, and a band of approximately 80–90 m immediately outside the grid received a double application of bait on both drops.

Fernbird transfers

In case the ground bait station network failed to protect the fernbirds, a second population was established on another island so that they could be re-introduced to Codfish Island if the Pacific rat eradication removed them from Codfish Island completely. The choices for islands were limited by the presence of either predators or other subspecies of fernbirds. Consequently the first transfer was to 12 ha Kaimohu Island south-west of Stewart Island. This transfer failed for unknown reasons, although it was thought

Table 1 Record of 416 bait stations on a 50m x 25m grid covering approximately 37 ha on Codfish Island.

Date of Check	No. of Stations Active ie bait still being taken.	% of Total Stations Active
17/8 416 bait stations loaded with 100 g Agtech pellets		
18/8	First Aerial drop	
18/8	65	15.6
19/8	90	21.6
20/8	111	26.6
21-22/8	111	26.6
23-24/8	19	4.5
23-24/8	Bait changed to 10 x 2 g baits	
25-26/8	20	4.8
27/8	Second Aerial drop	
28-29/8	10	2.4
30-31/8	0	0
Stations checked every 3 days until 24/9	0	0
24/9 Bait changed to 28 g “Contrac” wax block		

to be less than perfect prior to the introduction due to the island's size and vegetation types. As there were no other readily-available rat-free islands, the transfer team was then left with the only choice of preparing another island by removing predators from it.

The most suitable island for this was identified as Putauhinu (144 ha), which had a wide range of habitat including areas of pakahi (wetland scrub) similar to that preferred by the fernbirds on Codfish. This operation cleared the way for the transfer of 21 fernbirds from Codfish Island to Putauhinu in November 1997.

Safeguarding short-tailed bats

The other species identified as being at potential risk was short tailed bats. Various trials for both primary and secondary poisoning indicated that the risk was minimal although it could not be discounted entirely. Hence 50 bats were transferred to Ulva Island in an attempt to establish a second population of the bats. The transfer was not successful (B. Lloyd pers. comm.). Therefore holding sufficient bats in captivity for the duration of the project to safeguard the population was the only practical alternative.

Four purpose-designed "batteries" were constructed on Codfish Island and 386 bats were held for nearly three months. Only nine bats were lost up until the last week of the programme when 42 died in one event due to heat stress in one of the roost boxes. Even with this unfortunate event, the operation was still judged a major success by international bat husbandry standards. A team of 5-7 people was tasked solely with carrying out the bat protection and monitoring the wild population, and this investment of single task personnel is one of the main reasons for its success.

The wild population was also monitored during the operation using radio tracking and video monitoring of roosts.

RESULTS

Bait monitoring

The requirement for three fine nights following the bait drop was not met for this operation as there was 5mm of rain the night after the first drop. Despite this, the Agtech baits lasted longer than expected from the trials carried out in 1994 on Codfish Island. It was anticipated that most of the bait would have broken down/become unpalatable within about 20 days or 15 mm of rain, however baits in sheltered sites lasted several months and all the bait lasted long enough to do a more than adequate job. After the operation, bait was observed to breakdown at different rates in different habitats. The bait under the forest canopy broke down first, apparently because once it got wet it didn't dry out again very readily and so crumbled. However, the bait on the open pakahi areas lasted far longer, because even though it got wet more quickly, any wind or sunshine soon dried it out, forming a hard crust layer on the outside of the bait. This was particularly true for bait on moss, as the moss held the bait up slightly off the damper substrate.

Rat monitoring

There was minimal follow-up carried out after any of the three operations. Monitoring of Putauhinu relied largely on the muttonbirders, who spend up to eight weeks on the island each year and would soon have detected any rat sign if present. A trapping programme was put in place on Rarotoka with over 1000 trap nights not catching a single rat. This meant that in 1999 (after two years with no sign of rats), both islands were declared rat free.

On Codfish Island a student undertook a radio telemetry study of Pacific rat home range size and die-off during the eradication. Thirteen rats were radio collared and all were dead within seven days of the first bait drop. The information gained was useful to relay the timeframe of the die-off to senior managers and the media, but was not a guarantee of successful eradication.

For Codfish Island there was only one opportunity to carry out the eradication, due to the logistics of shifting kakapo, holding bats, and not having a budget for an immediate repeat. Intensive monitoring immediately post drop was neither feasible or warranted. Therefore there was no real advantage in spending a lot of money on learning the outcome (success or failure) straightaway as success could not be confirmed for two years anyway.

The initial proposal for post-operational monitoring for rats was to rely on the kakapo supplementary feeding stations that are scattered around the island – predominantly near the top of the island. These stations had acted as an attractant for rats in the past and there had been a trap network set up near them to reduce the interference from Pacific rats. The project team agreed that monitoring of these stations would be sufficient to monitor the success of the operation. However, after the return of the kakapo, the supplementary feeding was phased out temporarily by the kakapo management team. So a rat trapping programme was carried out in March 2000 with 180 traps being set in lines of ten, spaced around the island for a total of 1000 trap nights (I. McFadden pers. comm.). No rats were caught but the project team decided to wait until the kakapo supplementary feeding programme was restarted in September 2000 and monitor it for at least three months before declaring the eradication a success. This was done and the eradication was declared successful in December 2000.

There have been several possible, if unlikely, sightings of rats on Codfish Island since the eradication. Some of these have been discounted but others have required follow-up trapping which has not caught any rats.

Non-target monitoring

Bats on Codfish and fernbirds on both Codfish and Putauhinu were the only species actively monitored during the eradication programmes on the three islands. The decision not to actively monitor other species was based on the observations made as part of previous island eradications including Kapiti (Empson and Miskelly 1999). Monitoring of the previous eradications had shown few detrimental effects for non-target species following aerial application of brodifacoum poison. In addition, ecological monitoring designed to accurately show the results of the Pacific rat eradication would have been a large undertaking and very likely cost more than the eradications themselves.

If the changes shown on other islands are replicated, what can be expected is an increase in the diversity and abundance of indigenous species recorded. This is already occurring and is most obvious with an increase in insects, small birds, lizards, and seedlings of several plant species.

Bats

Monitoring of both the wild population and released captive bats did not show any observable loss and it is believed that there were at worst minimal individual losses, and certainly no observable effect at a population level. This task has been reported separately (Sedgeley and Anderson 1998; Sedgeley 1998, pers. comm).

Fernbirds

In November 1997, 21 fernbirds were transferred to Putauhinu and confirmation of their breeding the following summer meant that the Codfish Island eradication could go ahead. Follow-up checks on Putauhinu have shown that the fernbird population has continued to increase and expand its range (P. McClelland pers. obs.).

While not quantified, it appeared that most of the fernbirds on Codfish Island were killed in the poison operation, with very few being recorded for two years after the drop. However enough have survived to allow the population to build up rapidly and expand into most of its former range (P. McClelland pers. obs.).

Table 2 Birds found dead following Codfish Island bait drops on 18/8/98 and 27/8/98.

Species	mg/kg brodifacoum	Death probably caused by toxin	Date collected
Tui (*Prosthemadera novaeseelandiae*)	0.32	Yes	18/09/98
Tui	<0.02	No	No date
Blackbird (*Turdus merula*)	0.87	Yes	27/08/98
Thrush (*Turdus philomelos*)	1.0	Yes	24/09/98
Morepork (*Ninox novaeseelandiae*)	0.78	Yes	24/09/98
Kakariki (*Cyanoramphus spp*)	<0.02	No	11/09/98
Kakariki	1.4	Yes	01/09/98
Kakariki	0.03	No	30/08/98

Other species

Following the two bait drops eight individual birds were collected and analysed for the presence of brodifacoum. This suggested that brodifacoum poisoning killed individuals of five different species (Table 2).

General observations

The recovery of vegetation and fauna on Putauhinu and Rarotoka has been dramatic, although significantly faster and more obvious on Putauhinu (P. McClelland pers. obs.), with its better seed source and greater species diversity. There have also been major increases in the numbers of invertebrates, especially weta on Putauhinu and stag beetles (*Hemidorcus* spp.) on Rarotoka (P. McClelland pers. obs.). On Putauhinu many native bird species, including saddleback (*Philesturnus carunculatus*) and parakeets (*Cyanoramphus novaezelandiae* and *C. auriceps*) as well as the numbers of lizards and invertebrates have increased in numbers dramatically (J. Lee pers. comm.). Since the eradication, Stewart Island robin (*Petroica australis rakiura*), another species eliminated from Putauhinu by cats, have been re-introduced to Putauhinu and have rapidly increased in numbers (J. Lee pers. comm.).

The many years of uncontrolled grazing on Rarotoka with sheep and cattle has meant that vegetation recovery is far slower with a dense grass sward stopping many seedlings. A significant planting programme will be required to revegetate the island prior to considering re-introducing many species (P. McClelland pers. obs.).

DISCUSSION

The eradication on Putauhinu and also Rarotoka both removed the introduced predators from these islands and provided a suitable site for the second fernbird population. These eradications also gave the Codfish Island eradication planning and operational team a test run for such things as consents, weather forecasting, loading team setup, etc., which helped ensure that the latter operation went smoothly.

The longer lasting Wanganui No. 7 bait was suitable for Putauhinu and Rarotoka, with their limited non-target issues. The selection of the faster-deteriorating Agtech bait for Codfish Island was shown to be the correct choice, as all rats ate bait and non-target losses were minimal, even though the desired “three fine nights” after each bait drop did not eventuate.

While no quantifiable monitoring of the benefits of the eradications has been carried out, there have been a number of obvious benefits. These include finding invertebrates and lizards not previously recorded on those islands, noticeable increases in lizard numbers, and increases in numbers of some bird species.

The partial failure of the core area to protect all of the fernbirds it contained, is believed to be due to the bait lasting longer than expected. This meant that once any birds on the perimeter of the core area were killed by the bait, the birds in the centre expanded their territories into areas where bait was still available and were also killed. Whilst it was not possible to remove this risk, the impact may have been reduced by expanding the size of the core area, thus increasing the percentage of birds away from the perimeter. The partial failure of the core area to protect as many fernbirds as anticipated justified the additional costs involved in carrying out the eradication on Putauhinu, and the transfer of fernbirds to Putauhinu to establish a back-up population.

The three operations were successful largely due to the team approach used for the planning and field work, attention to detail by all personnel involved, forward planning, and using the information gained from previous eradications. The Island Eradication Advisory Group, a Departmental peer review group, ensured that these vital actions all occured.

The eradication of Pacific rat removes the last introduced predator on Codfish Island following the removal of Stewart Island weka *(Gallirallus australis scottii*) in 1984 and brushtail possum (*Trichosurus vulpecula*) in 1987 (A. Cox pers. comm.).

The removal of Pacific rats will allow greater regeneration of a number of trees, shrubs and herbs, including some rare species, which up until now were in part at least limited by rat predation of seed and seedlings (B. Rance pers. comm.). A range of both terrestrial and small seabirds will also benefit substantially from the removal of the sole remaining introduced predator, and it is predicted that their populations will increase significantly. Seabirds, eliminated by the rats and other predators, may naturally recolonise Codfish Island from neighbouring islands. Broad-billed prion (*Pachyptila vittata*) have been found in burrows on Codfish Island in 1999 (D. Scott pers. comm.), the first occurrence in over 60 years (M. Imber pers. comm.).

The removal of Pacific rats will allow the reintroduction of a number of terrestrial bird species presumed to have been eradicated by the rats, including robins (*Petroica australis*) and saddlebacks (*Philesturnus carunculatus*), and the introduction of other endangered species not previously found on the island but which require a safe haven, for example Campbell Island teal (*Anas nesiotis*). The removal of rats will also allow the resident impoverished invertebrate, reptile and small bird populations to recover. This programme has also created opportunities for introducing to Codfish Island threatened flora and fauna from other islands in the Titi Islands group, many of which hold remnant populations of threatened and endangered species (e.g. cloudy gecko (*Hoplodactylus nebulosus*)).

The next challenge is to ensure that these islands remain rodent free. Appropriate quarantine will require even more dedication than the eradications as it has to go on indefinitely with none of the glory or recognition of carrying out a successful eradication. The muttonbirders on Putauhinu have realised the value, both economically and to conservation, of having a rat-free island and are a leading example of rodent quarantine on a privately owned island. As Codfish is staffed and access is controlled by permit, quarantine is able to be strictly enforced, with strict standards being set for all visitors. Compared to Codfish, Rarotoka is relatively easy to access, is not inhabited, and hence presents a greater quarantine risk that will be managed by the owners with advice from DOC.

The eradication debriefs recommended that among other things:

1. The Department set up a national mechanism to record changes of bait and toxin formulation, so that managers can more readily attribute failure of any operation to a specific cause. This may be achieved by having a single national contact with toxin and bait suppliers and not accepting non-standard baits.
2. Managers do not set unrealistic goals as far as weather forecasting is concerned, as even Agtech bait lasted far longer than anticipated. However, there must be confidence that significant rainfall will not occur.
3. Detailed monitoring of potential non-target species in future operations should only occur when that species is at risk at population level and has not been monitored as part of a previous operation.

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Alien mammal eradication and quarantine on inhabited islands in the Seychelles

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Abstract During the period 1996-2000, eradication of five introduced mammal species (feral cat (*Felis catus*), rabbit (*Oryctolagus cuniculus*), ship rat (*Rattus rattus*), Norway rat (*R. norvegicus*) and house mouse (*Mus domesticus*)), was attempted on four inhabited islands, including three resort islands, ranging in size from 101–286 ha in the Seychelles group, Indian Ocean. Objectives were to avert extinctions of, and restore urgently-needed habitat for, localised threatened endemic animals and to facilitate ecological restoration in line with a national biodiversity strategy. Local political, economic and biological constraints meant that adaptations were necessary to traditional poisoning and trapping methods and regimes. Furthermore, since no rat-free island was available to which native animals at risk from primary and/or secondary poisoning might be transferred, it was necessary to maintain approx. 590 individuals of three threatened animal species in captivity for the three months of the eradication programme. Strategies and techniques developed, and some of the many challenges encountered in conducting eradication and quarantine programmes on inhabited, tropical islands are outlined, together with progress to date. One island (Bird) has been maintained free of rats and rabbits since their eradication in 1996. Two others (Denis and Curieuse) are now free of feral cats but have been recolonised by *Rattus rattus* since eradication attempts in 2000. The fourth (Frégate), was successfully cleared of *R. norvegicus* and mice in 2000, in time to avert extinctions of localised threatened endemic animals. These positive results will, we hope, inspire similar effort on other inhabited islands with high biological values or potential.

Keywords threatened species recovery; ecological restoration; eradication on inhabited islands; rodent eradication; cat eradication; Indian Ocean islands

INTRODUCTION

In 1964 when ship rats (*Rattus rattus*) irrupted on Big South Cape, a remote 939 ha island off southern Stewart Island, New Zealand, few anticipated the massive ecological impact, or the extent to which this event would shape future island conservation policy and practice both within New Zealand and beyond. Big South Cape was the final refuge for several endemic New Zealand animals – three birds, one bat and an unknown number of invertebrate species. Swift action by NZ Wildlife Service staff averted extinction of one bird, the South Island saddleback (*Philesturnus carunculatus carunculatus*), but all others were exterminated by the rats (Bell 1978; Merton 1978; Atkinson 1985). Although some conservation workers of that era recognised the threat rats pose to island biotas, many did not (Bell 1978; Galbreath 1993). The ecological collapse of Big South Cape was thus, both nationally and internationally, an important and timely lesson from which modern island rodent quarantine, contingency policies and protocols largely originate (Moors *et al.* 1989).

Thirty years on, in spite of major advances in both knowledge and eradication capability, and in the full glare of the international conservation spotlight, it is indeed remarkable that the biological tragedy of Big South Cape was very nearly repeated in the Seychelles. In 1995, Norway rats (*R. norvegicus*) reached Frégate Island, the Seychelles' last remaining rat-free island greater than 100 ha in area. Frégate is the principal refuge of two birds, three invertebrates and a mollusc endemic to the Seychelles, and supports the largest populations of six endemic reptiles. Five years were to elapse before any sustained eradication attempt was made, and this was driven largely by the negative impacts of rats on commercial tourism interests rather than threats to biodiversity conservation, such as the impending extinction of a giant tenebrionid (*Polposipes herculeanus*) (Parr *et al.* 2000). How, in this age of environmental awareness, could this situation arise, and how can we ensure prompt and effective action is taken next time an important conservation island is invaded by rats?

In this paper, we describe eradication campaigns against invasive mammals on four Seychelles Islands and protocols to prevent re-invasion.

The Seychelles

The Seychelles (Fig. 1) is an isolated archipelago of approx. 115 granitic and coral islands that occupy a land area of 445 km^2 and span 1200 km of the tropical Indian Ocean between 4^o and 8^o S. The four islands included in the eradication programme are within the "inner group" comprising the most northerly cluster of approx. 40, mainly granitic islands rising from a continental shelf of Gondwanan origin. The Seychelles had no indigenous human population. Europeans first discovered the group in 1609. Although known to other seafarers, such as Arabs, possibly as early as the 10th century (Benedict 1984), permanent human settlement began in 1770. Today, four large islands

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within the inner group hold almost all the human population of 76,500 with over 90% found on the largest island Mahé (152 km^2) (Benedict 1984).

Like other long-isolated oceanic islands, the Seychelles had no indigenous land mammals other than bats. Consequently, its animals and plants had few innate defences against such animals once introductions began in the late 18th century. Of surviving endemic birds, most have undergone a massive retraction in range and numbers (Stoddart 1984; Diamond 1985). For example, by the 1980s only approx. 20 Seychelles magpie-robins *(Copsychus sechellarum)* survived on a single island (Frégate); currently the Seychelles fody (*Foudia sechellarum*) exists only on four islands, three of them less than 50 ha in area; the grey white-eye (*Zosterops modestus*), now virtually extinct on the main island, survives elsewhere on just one approx. 60 ha island; the Seychelles brush warbler (*Bebrornis seychellensis*) was until the 1980s confined to one small island (Cousin, approx. 29 ha); and the black paradise flycatcher (*Terpsiphone corvina*) has a severely restricted range on only two islands. By 1997, introduced mammalian predators had colonised all but a few small islands, leaving a total of only 280 ha free of cats and/or rats.

Watson *et al.* (1992) suggested that the feral cat (*Felis catus*) was the cause of Seychelles magpie-robin extinctions on Aride and Alphonse as well as their serious decline on Frégate. It is, however, difficult to separate the effects of rats and cats because both species are present on most islands – cats were often introduced in an attempt to control rats.

The four inner-group islands of Bird, Curieuse, Denis and Frégate were the focus of recent attempts to eradicate introduced invasive mammal populations. Frégate has a small harbour with a wharf where small boats can tie up; the other islands are accessed by light fixed-wing aircraft or by dinghy ferrying goods and passengers from a launch standing offshore.

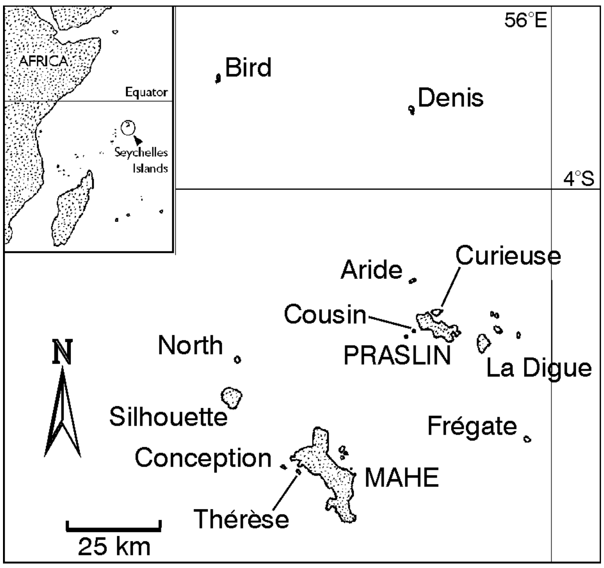

Fig. 1 Location of the “inner” (granitic) Seychelles Islands including Bird, Curieuse, Denis, and Frégate Islands, the focus of recent eradication projects.

Bird Island

Bird Island (101 ha) is a privately owned, low-lying, modified coralline sand cay 105 km northwest of Mahé. It is the northern-most of the Seychelles group. A tourist lodge built in the 1970s caters for up to 40 visitors with around 40 resident staff and their families. Although known internationally for its massive sooty tern *(Sterna fuscata)* nesting colony numbering about half a million breeding pairs, the island has no legal protective status. Ship rats reached the island in bundles of *Latania* fronds used for roofing thatch from Praslin Island (G.Savy pers. comm.) during construction of the lodge in the 1970s. Rats reached remarkably high densities during the sooty tern breeding season and were an ongoing serious problem for both the island’s management and resident wildlife. Rabbits (*Oryctolagus cuniculus*) and mice (*Mus domesticus*) were present by the early 1900s.

Curieuse Island

Curieuse (286 ha) is a granitic island situated 52 km northeast of Mahé and 1 km off the northwestern coast of Praslin. The island rises steeply to a central ridge that reaches 172 m at its highest point. Coastal plateaux and lower slopes are heavily forested. The island is renowned for its outstanding biological values which include some of the least modified vegetation associations to be found in the Seychelles, as well as a number of threatened endemic plants and animals. From 1833–1965 the island was occupied by a leper colony. It has since been designated part of Curieuse Marine National Park and is managed by the Marine Parks Authority with about seven rangers in residence. Feral cats, ship rats, and mice were present prior to the current eradication attempt. Since the island is relatively large and already conservation estate, its potential for re-introduction of threatened endemics has long been recognised. Eradication of cats and rats was considered essential if the island was to function as an effective reserve for indigenous animals and plants.

Denis Island

A 143 ha, privately owned coralline island situated 95 km north of Mahé, Denis is low-lying; its highest point being only ~4 m above sea level. The island is forested but for the airstrip and clearing associated with a tourist lodge near the northern coast. Most natural vegetation was cleared last century and the island managed as a coconut *(Cocos nucifera*) plantation until the 1950s, when the plantation was abandoned. Wild coconut is now dominant both in the canopy and understorey, however significant native forest remains. Feral cats, ship rats and mice have long been present. Although the island’s natural values have been degraded, it has potential for the conservation of indigenous plants and animals. The island has no conservation protective status.

Frégate Island

Frégate (219 ha), is a privately owned granitic island situated 55 km east of Mahé. Apart from a 35 ha plateau near the north-eastern coast where a tourist lodge, staff houses, cultivations, airstrip and small boat harbour exist, the island is forested. The highest point, Mont Signale, is 125 m above sea level. All natural vegetation was cleared last century and the island was managed as a copra plantation until the 1950s. The plantation has since been abandoned and a forest association dominated by coconut, cashew (*Anacardium occidentale*), sandragon (*Pterocarpus indicus*), cinnamon (*Cinnamomum* sp.), citrus (*Citrus* sp.) and other formerly-cultivated species now covers the greater part of the island. Feral cats were eradicated in the 1980s by C. R. Veitch and V. Laboudallon with local assistance (Watson *et al*. 1992).

The first Norway rat was reported on the island on 11 September 1995 (M. Rands pers. comm.; Jones and Merton 1995; Merton 1996; Thorsen *et al.* 2000). By July 1998 the population had irrupted and dispersed throughout the island (Merton 1999). Frégate is an important refuge for many endemic animal species considered highly vulnerable to rat predation. Prior to the 1990s the Seychelles magpie-robin was confined to Frégate, which still supports the largest population of magpie-robins, along with the largest of only four remaining Seychelles fody populations. Frégate provides vital habitat for at least six endemic reptile and several invertebrate species, and is the sole refuge of a giant endemic tenebrionid (*Polposipes herculeanus*), and endemic molluscs (*Pachnodus fregatensis* and *Conturbatia incisa*) – the latter, a genus known only from Frégate. The island has no formal conservation status.

Ecological collapse, similar to that of Big South Cape Island in the 1960s, seemed likely on Frégate. Merton (1996) therefore recommended that an immediate attempt be made to eradicate the colonising rats and that ongoing monitoring of key endemic animals, such as the giant tenebrionid, be carried out to establish population baselines and future trends. DM recommended that other islands with conservation potential be cleared of rats at the same time to increase the severely restricted rat and cat-free habitat remaining for vulnerable Seychelles endemics and so the Frégate operation might benefit from economy of scale.

Thorsen *et al.* (2000) describe initial attempts to contain and eradicate rats while their population was small and localised during their early colonisation of Frégate. The project was abandoned when several Seychelles magpie-robins disappeared and one was found moribund with symptoms consistent with rodenticide poisoning. BirdLife Seychelles instigated regular monitoring of some threatened endemics in early 1996. During the period March 1996–November 2000, the adult population of the giant tenebrionid declined by about 80% (Parr 1999). In the 1999/2000 breeding season, 19 magpie-robin fledglings disappeared within the first few days after leaving the nest (Millett pers. comm.). Both events were linked to predation by rats (Parr *et al*. 2000).

Background to eradication campaigns

In November 1995, at the owner's request, DM visited Bird Island to study the feasibility of eradicating rats and rabbits. The successful outcome on Bird (Feare 1999), coupled with the arrival of Norway rats on Frégate Island in 1995 (Merton 1996; Thorsen *et al.* 2000) sparked interest in ecological restoration of further islands.

In 1997, the Seychelles Ministry of Environment and Transport (MET) resolved to eradicate alien mammals from Curieuse Island as part of its ecological restoration programme, and to coordinate similar work on other islands (Ministry of Environment 1997). The Conservation Division within MET funded DM, via a Dutch Trust Fund grant, to study the feasibility of eradicating rats and feral cats from Curieuse, Denis, Conception and Thérèse Islands, and rats from Frégate Island. DM, assisted by VL and other MET conservation staff, visited the five islands in July – August 1998 and submitted a report and eradication proposal to MET (Merton 1999).

Three islands (Curieuse, Denis and Frégate) eventually formed the basis of the eradication plan adopted and implemented by the Seychelles Government, in collaboration with management of Frégate and Denis Islands and BirdLife Seychelles (Merton 2001). The project was coordinated by MET, funded by island owners and management, together with a grant from the Dutch Trust Fund, and organised by DM, who led the eradication team comprised of New Zealanders and Seychellois.

Goals of the proposed programme were:

- to prevent global extinction of endemic Seychelles animals confined largely or entirely to Frégate Island;
- to provide urgently needed habitat, free of alien mammalian predators, for the expansion in range and numbers of threatened endemic animals whose relict populations currently occupy dangerously restricted ranges – an essential first step towards ecological restoration in line with a national biodiversity strategy;
- to enhance these island environments from a human use perspective including eco-tourism. These benefits were important because the eradication and subsequent rodent quarantine could not proceed without the support of local communities and government, and funding from island owners.

To meet these goals the following objectives had to be achieved:

- rodent quarantine and contingency plans developed for, and accepted and implemented by, management and staff on each of the islands;
- local conservation staff trained in eradication techniques;

- rats eradicated from Curieuse, Denis and Frégate Islands, and cats from Denis and Curieuse. (While not a core objective, it was hoped that mice would also be eradicated, due to their potential impacts as predators of invertebrates and reptiles if they reached high densities after the removal of rats and cats.)

A ground-based operation was less practicable on these islands due to their larger size, dense vegetation cover and, in the case of Curieuse and Frégate, more rugged topography. Some economy of scale was achieved through incorporating the three islands into a single eradication operation.

BIRD ISLAND FEASIBILITY STUDY

In November 1995, DM assessed densities and distribution of target species on Bird Island (ship rats, rabbits and mice); the most preferred and appropriate bait and means of presentation; non-target species at risk and how these risks could be managed; optimum timing of poison application; logistics, costs, and resourcing requirements. The ability to implement and sustain quarantine and contingency measures to prevent re-introduction of rats was assessed through discussions with the owner and island staff.

Rodent indices and population data were obtained through cage trapping. Twenty-three mesh cage traps, baited with grilled coconut pieces, were sited at 50 m intervals along an index line which took in open terrain, forest and part of the sooty tern colony. Traps were opened on specific nights only. Rat index trapping results (using the method of Cunningham and Moors (1996) to obtain a standardised index of rat abundance expressed as captures per 100 corrected trap nights (CTN)), age, sex, weight, and breeding status were recorded (Appendix 1).

Index trapping in November 1995 indicated that rats were present at high densities (26.5 rats/100 CTN) (Appendix 1). Their ecological impact was evident from the high level of rat predation on nesting sooty terns during the breeding season (S. Robert pers. obs.). Mice were seen only occasionally throughout the island but could be expected to become more numerous and problematic should they survive the eradication attempt. Rabbits were rarely seen and were localised on the airstrip.

Eight commercially-available rodenticide bait types, of which three were available in non-toxic form, were compared for acceptance by target and non-target animals, and durability in a tropical environment (Table 1). Bait preference by rats was ranked by placing a measured weight of each bait under each of eight stations (upturned buckets with a 50mm diameter hole cut in the side) overnight and weighing how much of each type remained the following morning. For each station, there was a paired control comprising a similar selection of baits placed in the open where they were vulnerable to disturbance and removal by the full range of target and non-target species. This process was repeated for five nights. Land crabs (*Cardisoma* sp.) often removed 100% of baits from open plots in coastal sites. Preference trials were also conducted with caged rats. The three non-toxic bait types were offered at a feeding table traditionally visited by a range of bird species. Bait preference to birds was assessed by direct observation. To assess weathering, baits placed under 1 m x 1 m wire 10mm mesh covers, which excluded all animals other than small invertebrates, were monitored for signs of breakdown, mould, and removal by ants or cockroaches.

Wanganui No. 7 standard 12 mm pelleted bait (20 ppm brodifacoum; Animal Control Products Ltd (ACP), Wanganui, NZ) was chosen for hand broadcasting to target rats, mice, and rabbits. This was more acceptable to rats than other pelleted baits trialed, was more durable, less prone to ant damage, and also less acceptable to non-target species – presumably because it was green-dyed to deter birds (Caithness and Williams 1971) (Table 1). Talon 50WB™ wax blocks (50 ppm brodifacoum; 18 g per block) were sufficiently acceptable and durable to be maintained in bait stations throughout the programme, which was to extend through the “wet season” (November–March).

Non-target species at risk included ruddy turnstone (*Arenaria interpres*) accustomed to feeding at a bird table, cattle egret (*Bubulcus ibis*), introduced Madagascar turtle dove (*Streptopelia picturata*), barred ground dove *(Geopelia striata)*, Madagascar fody *(Foudia madagascariensis)*, Indian mynah (*Acridotheres tristis*), and domestic poultry, pigs and a dog, as well as young children. Poisoning of individuals of more numerous, introduced, ground-feeding species was considered inevitable, but acceptable to the owner and unlikely to have any significant or lasting impact on populations. The potential risk to reptiles was considered to be low (Merton 1987). However, as a precautionary measure, it was agreed that all three Aldabran giant tortoises (*Geochelone gigantea*) would be held in captivity during the campaign (i.e. from before the first poison application until after all baits had been removed).

BIRD ISLAND ERADICATION CAMPAIGN

The one child under four years old was removed from the island for the duration of the programme. Domestic livestock, a dog, and three Aldabran giant tortoises were confined to prevent access to baits. Domestic poultry and pig feeding regimes were modified to prevent rats gaining access to stock foods, and protocols were implemented to prevent rats gaining access to stored foods or refuse.

Parallel transect lines to provide foot access were cut at 50 m intervals across the entire island and marked with tape. Index trapping, using the same method as in November 1995, was carried out before and during the November 1996 eradication operation.

The poisoning operation began on 30 October 1996, immediately after the majority of the sooty tern chicks, a major food source for the rats, had left the island, but before the NW monsoon and associated wet season began in late November. The tourist lodge remained open throughout.

Three to five Talon 50WB wax blocks were placed in each of 800 bait stations sited at 50 m intervals along each transect line. Bait stations were made from 1 *l* plastic drink bottles with one end cut off, or from 400 mm lengths of 110 mm PVC pipe. To reduce bait consumption and scattering by non-target species, especially skinks, masking tape was stretched across the lower portion of station openings. Hermit crabs, which were especially abundant in the coastal zone, were largely excluded by elevating bait stations 1-2 m above the ground; this was achieved by tying each bait station to the central rib of a cut coconut palm frond lodged horizontally in the fork of a low growing palm.

Approximately 60 kg of Talon wax block bait was laid in the first loading of bait stations. Four days later, each was replenished with 2 blocks of bait because many bait stations were empty after the first night. It was apparent at this point that many rats were already dead because most of this second pulse of bait remained after three nights.

Table 1 Rodent baits trialed on Bird Island, November 1995 after five days exposure and 10 mm rainfall. (The three pellet types produced by Animal Control Products (ACP), N. Z. were available in both toxic and non-toxic forms; the remaining six products were available only as toxic baits).

Trade name	Active ingredient	Exposed on damp forest floor	In covered bait stations	Comments
"Tornado" blue cubes South Africa	Difenacoum 100ppm	No change	No change – become soft during heat of day, but maintain shape	Excellent durability, no bird/ant interest, little loss to crabs
"Klerat" dark blue wax blocks ICI, UK	Brodifacoum 50ppm	No change	No change	Excellent durability, no bird/ant interest, little loss to crabs
"Storm" Light blue–pillow shape, 16.5 g Shell, UK	Flocoumafen (50ppm)	No change	No change	Excellent durability, no bird/ant interest, little loss to crabs
"Ditrac" blue wax blocks	Diphacinone (50ppm)	No change	No change	Very good durability; Some ant damage, little loss to crabs
"Talon 50WB" wax blocks, 18 g ICI, NZ	Brodifacoum 50ppm	Little change Melt readily in direct sunlight	No change Soft during day	Good durability; relatively low melting point a disadvantage, little loss to crabs
"Wanganui rodent pellets", cereal, 7mm diam. ACP, New Zealand	Non-toxic (normally Brodifacoum 20ppm)	Soft, crumbling, heavily attacked by ants, many partially buried	Considerable ant damage to some	Good durability, ants and crabs a serious problem, break down rapidly in rain
"Wanganui No. 7" standard 12 mm diam. cereal pellets = "PestOff 20R" ACP, NZ	Non-toxic (normally Brodifacoum 20ppm)	Swell slightly under humid conditions, no crumbling or erosion, colour fading, 10% partially buried by ants	No change, little interference by ants, some mould on one	Excellent durability except in very wet, relatively little interest by ants and birds; significant loss to crabs
"Agtech R5" pellets ACP, NZ	Non-toxic (normally Brodifacoum 20ppm)	Swollen, eroding & crumbly	Significant erosion of surface, ant and structural damage	Poor durability, due to erosion by climate and ants, significant loss to crabs
Wax candle (gnaw-stick)		Melt and droop in heat	Melt and droop in heat	Melt badly, rodent sign confused by that of crabs and cockroaches

Stations were subsequently restocked at monthly intervals. Monthly refilling continued until April 1997, when all but 12 designated permanent bait stations were removed. In total, 200 kg Talon 50WB was used. This method facilitated ongoing poisoning of any rats that survived into the wet season (when it was not feasible to broadcast pellets since they would break down rapidly under the wet conditions) and provided more targeted delivery of toxin over an extended time frame (considered important for eradication of mice).

To intensify the initial knockdown of all three target species, one tonne of Wanganui No. 7" pelleted bait was broadcast by hand in two pulses at 10 day intervals (Nov. 8-12 and 18-20) (Appendix 2). The operator stopped every 25 m along each transect line to cast one measured cupful (approx. 100g) to the north, south, east, and west, and deposit a fifth at his/her feet, thus achieving a coverage of 4-5 kg/ha. Higher concentrations of bait were applied along coastal strips, where bait loss to land crabs was highest, and in areas of dense cover, in and under buildings and in ceilings. Lower concentrations of bait were laid in open areas of dry mown grass (approx 30 ha), which supported rabbits but few rodents.

Capture rates of rats prior to the first application of pellets reached 141 rats/100 CTN (Appendix 1). This high population density may have been associated with the unusually late departure of sooty terns from the breeding colony. Index trapping also confirmed that some breeding was in progress. It was therefore important to ensure that baiting catered for juveniles initially confined to nests or denied access to baits by dominant adults.

Nine days into the poison campaign, rat indices had dropped to 15.2/100 CTN and the majority were juveniles. No rats were caught when traps were again opened 16 and 17 days after poisoning commenced. By the third week there was no sign of live rats, mice, or rabbits.

Records were kept of any dead non-target species found but carcasses were not analysed to confirm cause of death. Non-target deaths detected (Table 2) included turnstone, cattle egret, Asiatic whimbrel *(Numenius phaeopus variegatus)* and introduced Madagascar fody, Indian mynah, Madagascar turtle dove and barred ground dove. Although Seychelles skinks were seen feeding on rain-softened pellets, there was no observed mortality.

Follow-up monitoring included regular examination of Talon WB baits in permanent bait stations, and checks of the sooty tern colony during subsequent breeding seasons for signs of rat predation on eggs and chicks. While the eradication of rats and rabbits proved to be successful, mice were seen in the lodge in early 1998, having either survived the eradication attempt or been accidentally reintroduced. Mice are now widespread and abundant on Bird Island.

CURIEUSE, DENIS AND FREGATE ISLAND CAMPAIGNS

The feasibility study carried out in July-August 1998 (Merton 1999) indicated that rats were at a relatively-high density on all three islands (Appendix 1) and that there was no evidence of more than one rat species on each island. An assessment was made of non-target species at risk from poisoning and we established mitigation measures, including aviaries and enclosures. Discussions with stakeholders, including island owners and staff, addressed optimum timing of poison application, logistics, costs, resourcing requirements and, most importantly, the ability to implement and sustain quarantine and contingency measures to prevent re-introduction of rats. Appendix 3 provides an example of the recommended protocol for Frégate Island. Timing of the operation, starting in early June rather than July/August, was dictated by the seasonal low in tourism rather than biological factors. An operation at any other time would have necessitated closure of the Denis and Frégate Island resorts in the height of the tourist season, with the loss of considerable revenue.

On Frégate Island, 39 magpie-robins and 330 Seychelles fodys were taken into captivity before the poison operation began in June 2000 and were held in rat-proof enclosures until baits were no longer available (11 weeks). A total of 215 Aldabran giant tortoises were also held captive on the three islands (see Table 2). In addition to the measures taken on Bird Island to minimise impacts on other non-target species, it was necessary during each aerial drop to protect aquatic fauna (e.g. endangered, endemic freshwater fish and terrapins) by covering ponds with polythene film. Roof water catchment down-pipes were disconnected and water tanks covered. Frégate giant tenebrionids had previously been established in captivity at London Zoo, U.K.

Livestock feeding regimes, human and livestock food storage and refuse disposal protocols were implemented on

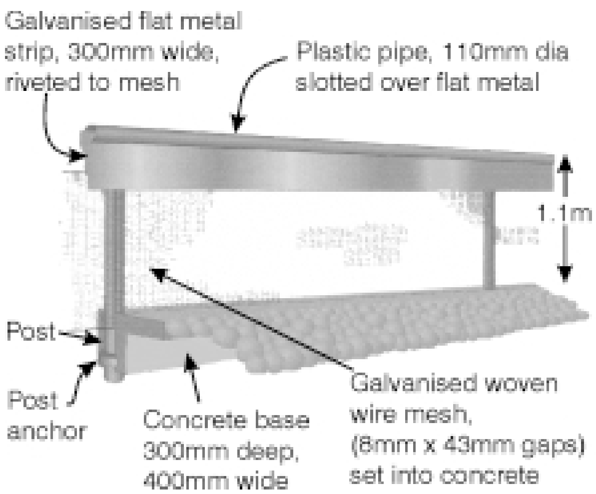

Fig. 2 Frégate Island Harbour rodent fence.

each island to deprive rats of alternate food sources, and rodent quarantine and contingency plans were put in place (Appendix 3). On Frégate Island, a rodent-proof harbour fence, designed by DM in consultation with MET and Frégate Island management, was constructed by Frégate staff (Fig. 2).

To determine when it was safe to release the several hundred threatened animals from protective confinement on Frégate Island, bait degradation was monitored within six exclosures. These exclosures were sited in representative vegetation types, at different elevations, aspects and exposures, and were designed to protect baits from interference by all animals other than invertebrates. Each exclosure comprised a rigid 1 m x 1 m wire bird-mesh (10 mm x 10 mm mesh size) cover 10 cm high. Twenty-five pellets were placed on the ground beneath each cover on the same day that bait was applied to the island. Pellets within each exclosure were counted and the number and condition recorded at least weekly thereafter until all pellets had broken down to such an extent that they were no longer recognisable and no longer posed a risk to wildlife.

Three 10 m x 10 m (100 m^2) open quadrats were established on each of Denis and Curieuse, and four on Frégate, in order to sample bait densities immediately after each aerial application and subsequently to monitor the rate of bait consumption/loss to both target and non-target species. So far as practical, sites were selected in different vegetation associations, and at different elevations, aspects and exposures. Each quadrat was measured using a tape measure, and wooden or metal stakes 40 cm in length were driven into the ground at each corner. Fine white string was then stretched and tied between stakes. For ease of counting, each quadrat was sub-divided into quarters using further stakes and string. Pellets falling within quadrats were counted immediately following aerial bait application and, where practicable, daily thereafter until all pellets had either disappeared or broken down to such an extent that they were no longer recognisable (Fig. 3).

A helicopter (Bell Jet Ranger) was hired from Helicopter Seychelles Ltd for each of the seven rodent bait applications. A differential global positioning system (DGPS) brought from New Zealand was installed into the helicop-

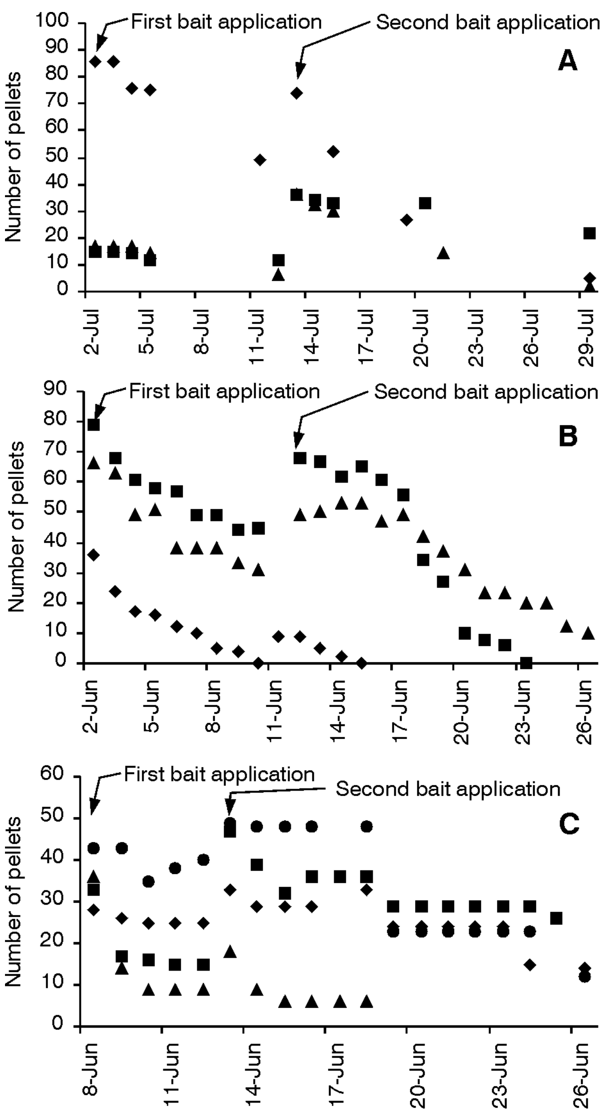

Fig. 3 Pellet density and rate of consumption or loss from 10 m x 10 m quadrats following aerial baiting on three Seychelles islands in the year 2000. The number of pellets in each quadrat can be translated to kg of bait per ha in the table:

Pellets/100m^2	10	20	30	40	50	60	70	80
kg/ha	2.5	5	7.5	10	12.5	15	17.5	20

The upper graph (A) for Curieuse Island shows: Quadrat 1 (diamonds) at Anse St Jose, sand beneath open mature coastal plateau forest; Quadrat 2 (squares) at Badamier Saddle, exposed, arid, hillside (65 m a.s.l.), eroded granite with sparse stunted tussock and scrub; Quadrat 3 (triangles) at Anse Badamier, damp forest floor beneath closed canopy of mature Badamier/Takamaka forest, 100 m from coast.

The centre graph (B) for Denis Island shows: Quadrat 1 (diamonds) in coastal *Scaevola* scrub, with high density land crab population; Quadrat 2 (squares) in mature forest near the centre of the island; Quadrat 3 (triangles) in open, close-cropped, grassland 50 m from coast.

The lower graph (C) for Frégate Island shows: Quadrat 1 (diamonds) at Au Salon in mature sandragon forest; Quadrat 2 (dots) at Grande Anse in open coastal cultivation with sparse, mature coconut; Quadrat 3 (triangles) at La Cour, mowed grass, coastal plateau near airstrip; Quadrat 4 (squares) at Gros Bois Noir in mature sandragon forest, inland plateau 75m a.s.l.

ter before each baiting. Prior to each bait drop the DGPS differential unit was set up on some nearby vantage point, taking its power supply from the battery of a tractor or other vehicle, the motor of which had to remain running throughout the baiting operation in order to cope with power demands. The helicopter was flown by an experienced New Zealand pilot with an "agricultural rating". Before the initial aerial baiting on each island, the island's boundary was flown and coordinates logged into the computerised DGPS system. Parallel transects at 40 m spacing were generated on the GPS screen. The pilot then overflew these transects to evenly distribute baits from a motorised spreader hopper with an 80 m swath, slung beneath the helicopter. Thus, each swath overlapped 50% with the previous swath, so ensuring complete coverage. The spreader hopper and bait were imported from New Zealand.

The PestOff 20R pelleted rodenticide bait (20 ppm brodifacoum; ACP) used was identical to the Wanganui No. 7 pellets hand broadcast on Bird Island in 1996. Aerial applications were broadcast in two pulses with an 11-day interval on Curieuse and a nine-day interval on Denis. Three pulses, five and 24 days apart, were applied to Frégate Island (Appendix 2). An additional pass was made over the coastal zones of each island to compensate for anticipated heavy bait loss to crabs, with the intention that each 80 m coastal strip would receive a 50% heavier bait application rate than elsewhere.

Rat indices and non-target mortality were monitored as on Bird Island, except that the traps were set and checked daily without interruption before and during the campaign. The proportion of reproductively-active females (pregnant or lactating) present in May-June 2000 ranged from 5.8% on Curieuse (among 34 adult females caught, two were pregnant and none were lactating) to 66% on Denis (where only three adult females were caught, of which two were lactating). On Frégate, 38% were reproductively active. Catch rates plummeted immediately after the first aerial baiting. For example, on Denis and Frégate, no rats were caught after the first baiting, while on Curieuse, the trap catch rate over the first four nights following the initial bait application was 4/100 CTN and no rats were trapped subsequently (Appendix 1).

Checks of the 100 m^2 quadrats within hours of each aerial drop, before the nocturnal land crabs had opportunity to remove baits, indicated that bait application varied widely from the intended delivery rate of 12 kg/ha for inland sites and 18 kg/ha for coastal strips. However, even with bait applications of 10 kg/ha onto coastal quadrats with high density crab populations, baits were available at >2 kg/ha for at least four nights after each bait drop (Fig. 3). On all three islands, a significant amount of bait was also taken by ants, cockroaches, and introduced doves of two species.

After five days of exposure, pellets within Frégate Island exclosures were baked hard by the sun and upper surfaces had faded to a pale straw colour. Between seven and 12 days, ants had significantly eroded and partially buried many baits, and white mould had developed. After 12 to 20 days of exposure, most baits were encased in black mould. Those baits remaining above the surface after 25 to 30 days were heavily eroded and were hardly recognisable. By 44 days, no recognisable pellets remained and the exclosures were removed.

At least eight cats are believed to have died as a result of brodifacoum poisoning on Curieuse and Denis Islands. Three cats on Curieuse and one on Denis were found dead 14 days after the first rat bait application and before the first 1080 poison was laid for cats. Four others on Curieuse disappeared at this time. Some of these deaths are likely to have been due, at least in part, to secondary poisoning through eating poisoned rats. However, with a brodifacoum LD50 of 25mg/kg it is doubtful that cats would have been physically capable of consuming sufficient poisoned rats during this period to constitute a lethal dose (W. Simmons pers. comm.), and we suspect that in the absence of rats, these cats consumed rain-softened rat bait and died from primary poisoning.

Cat eradication commenced one week after the second rodent bait applications on Curieuse and Denis Islands (Appendix 2), the rationale being that cats deprived of their primary food source (rodents) would be more susceptible to poisoning and trapping. Also, it was important to minimise the use of 1080 toxin on these inhabited islands (Eisler 1995). Rats would have consumed much of the 1080 if it had been applied earlier, and bait aversion may have occurred if sub-lethal doses had been consumed.

Thirty cat bait feeding stations (modified "Philproof" mini bait stations; ACP) had previously been established near refuse dumps and other traditional cat feeding sites on Denis, and 70 stations were in place on Curieuse. These were stocked for the first week with non-toxic cat bait (PestOff non-toxic pelleted chicken meal cat bait; ACP), which was then replaced with toxic bait of identical type containing 0.1% 1080 (PestOff 1080 chicken meal cat bait). Some bait was also laid in selected natural sites in areas remote from human habitation.

Bait presentation had to be modified due to low acceptance by cats on Curieuse and Denis (in contrast to high acceptability by feral cat populations elsewhere (Morgan *et al.* 1990)). Satisfactory acceptance was achieved by grinding the 1080 cat pellets and mixing the resulting powder at a ratio of one part ground pellets to five parts by weight of canned tuna in vegetable oil. Two lethal doses were contained within 10 g of this mixture. Baits were monitored and replaced daily for 2–3 weeks. The first cat deaths attributable to 1080 poison were discovered on 26 June on Denis Island (three days after the first baiting) and in late July on Curieuse, where 1080 poisoning began on 22 July.

Cat trapping began one week after toxic cat baits were first laid. Lanes Ace and Victor 1.5 leg-hold traps were spaced at 100 m intervals along tracks and baited with (non-toxic) canned tuna. Ninety traps were deployed on

Table 2 Estimated mortality in non-target organisms resulting from rat eradication using brodifacoum, mitigating measures, and impacts. No non-target mortality is known to have resulted from use of 1080 for cat eradication. (Table adapted from Parr *et al.* 2000).

Non-target species	Status	Distribution	Mitigation measures	Estimated mortality
Pets and livestock	Introduced	All islands	Housed and penned	None
Seychelles magpie-robin *Copsychus sechellarum*	Critically endangered endemic	Frégate	Captive management 39 (100% of population)	None
Seychelles fody *Foudia sechellarum*	Vulnerable endemic	Frégate	Captive management 330 (50% of population)	7.5% of captive population (n=25); no observed mortality in free-living (50%)
Turnstone *Arenaria interpres*	Non-threatened migrant	All islands	Use of green-dyed bait and low (20ppm) brodifacoum loading.	Bird 12 (50%) Curieuse 5 (25%) Denis 25 (80%) Frégate 30 (90%)
Asiatic whimbrel *Numenius phaeopus variegatus*	Non-threatened migrant	All islands	Use of green-dyed bait and low (20ppm) brodifacoum loading	Bird 2 (20%)
Madagascar turtle dove *Streptopelia picturata*	Hybrid of introduced/ native stock	All islands	Use of green-dyed bait and low (20ppm) brodifacoum loading	Bird 20 (30%) Curieuse 20 (10%) Denis 80 (40%) Frégate 200 (80%)
Barred ground dove *Geopelia striata*	Introduced	All islands	Use of green-dyed bait and low (20ppm) brodifacoum loading	Bird 75 (50%) Curieuse 20 (40%) Denis 150 (40%) Frégate 300 (80%)
Indian mynah *Acridotheres tristis*	Introduced	All islands	Use of green-dyed bait and low (20ppm) brodifacoum loading	Bird 30 (70%) Curieuse 20 (40%) Denis 100 (60%) Frégate 25 (50%)
Madagascar fody *Foudia madagascariensis*	Introduced	All islands	Use of green-dyed bait and low (20ppm) brodifacoum loading	Bird 80 (50%) Curieuse 20 (40%) Denis 50 (40%) Frégate 200 (70%)
Cattle egret *Bubulcus ibis*	Native	All islands	Use of green-dyed bait and low (20ppm) brodifacoum loading	Bird 5 (50%)
Aldabran giant tortoise *Geochelone gigantea*	Introduced	All islands	Penned – Bird 3 (100%) Curieuse 70 (60%) Denis 5 (100%) Frégate 140 (90%)	Curieuse 1 Frégate 2 (No toxin-related mortality)
Skink and gecko spp	Endemic	All islands, especially Frégate	None	No observed mortality
Invertebrates	Endemic	All islands, especially Frégate and Curieuse	*Ex situ* populations established for certain species	No observed mortality

Curieuse, and 52 on Denis Island. Local conservation staff were trained in eradication techniques. Cat trapping and poisoning has since been continued by Division of Environment, MET staff on Denis, and MPA staff on Curieuse, working on a cycle of two weeks on and two weeks off. This regime continued through 2001 until eradication was achieved.

No formal measures were put in place to detect rat and cat survival because we considered it unlikely that any individuals could escape detection for long on inhabited islands. In effect, PestOff Rodent block baits within 180 permanent rodent bait stations on the islands would serve to indicate the presence of any rodent.

The capture and captive management of Seychelles magpie-robins and Seychelles fodys on Frégate by BirdLife Seychelles staff, and of Aldabran giant tortoises by Frégate Island Ltd staff on Frégate and Marine Parks Authority staff on Curieuse, was an outstanding success (Table 2). Avicultural knowledge and capability advanced enormously. Not only did magpie-robins breed successfully during three months in captivity, but chicks were artificially hatched, hand-raised and fostered between nests for the first time (Millett *et al.* 2000). All tortoises and Seychelles fodys were released in late July-early August, and the majority of the magpie-robins were released in mid August 2000.

The successful eradication of rats and mice from Frégate Island was confirmed in June 2002, 24 months after the eradication campaign. A mouse, which apparently arrived with cargo, was captured and killed on 27 September 2001 but there has been no sign of mice since (Millett and Shah 2001b). The absence of any rat sign, as of June 2002 (Millett pers. comm.), is encouraging. However, lack of commitment to the ongoing implementation of rat abatement measures to a sufficient standard (Millett *et al.* 2000; Climo 2001) remains the greatest challenge and continues to jeopardise the long-term success of the campaign.

Unfortunately, ship rats were discovered by BirdLife Seychelles staff on Denis and Curieuse Islands in August 2001, and have become widespread and abundant on Curieuse (Millett and Shah 2001b). Since initial reports indicated relatively small and localised populations, it is suspected that ship rats came ashore once again with building materials. In late 2001 it was also discovered that mice had re-invaded Denis Island or survived the eradication attempt there.

The last cat was trapped on Curieuse in February 2001, and the last two cats were destroyed on Denis Island between July and September 2001. There has been no evidence of cats surviving on either of the two islands since (Millett and Shah 2001b).

DISCUSSION

The biological and conservation benefits of eradicating alien pest animals from uninhabited or sparsely-inhabited islands have long been recognised. However, the practicability of permanently removing such pests (especially rodents) from oceanic islands supporting human settlement and/or development, together with information on any enduring ecological benefits, appear largely unknown. Eradication and, in particular, effective rodent quarantine on such islands has generally been considered impractical – if not impossible. Most rodent eradications to date have involved uninhabited conservation estate. However, the majority of the world's half million islands have no formal conservation status and are inhabited. Many have biological values or potential and warrant ecological restoration – including removal of invasive animals.

Successful outcomes on Bird and Frégate Islands have shown that rat eradication and quarantine on resort islands within the Seychelles is both feasible and beneficial, bringing immediate economic and biological benefits.
For instance, on Frégate Island:

- Eradication efforts appear to have come just in time to avoid extinctions;
- There has been no recorded mortality among dependent recently-fledged magpie-robins since the rat eradication on Frégate in June 2000 – a stark contrast to the loss of 19 (virtually all) newly-fledged young in the year prior to rat eradication (Millett pers. comm.);
- Production of fruit and vegetables, free of rat damage, for the lucrative local resort market is at last a reality.

On Bird Island:

- Following eradication of rats in 1996, Feare (1999) reported that "common noddies (*Anous stolidus*) have begun nesting successfully on the ground and turtle doves (*Streptopelia picturata*), many showing characteristics of the endemic race *rostrata*, became numerous; they had not been seen on the island since 1973";
- Predation of sooty tern eggs and young by rats, previously widespread, has ceased.

The project has helped advance our knowledge and confidence in eradication capability. For instance:

- Eradication of rats is feasible while rats are breeding;
- Cats and rats can be effectively poisoned and trapped in the presence of massive, alternative food sources such as those provided by colonial breeding seabirds (Denis and Frégate), fruit and produce (Bird, Frégate) and kitchen refuse (Denis and Frégate);
- Rat eradication is practicable in the presence of high-density land crab, hermit crab, ant and cockroach populations, such as on Bird and Denis Islands, and coastal zones of Curieuse and Frégate Islands. Following each aerial bait application, bait removal by crabs proved less of a problem than anticipated from bait preference trials on these islands. The initial trials likely over-estimated the degree of bait interference because crabs were converging on a limited food source avail-

able at few sites whereas, during the eradication campaigns, baits were available throughout their entire habitat. This was fortuitous because the amount of bait actually delivered within some open quadrats was significantly less than calculated bait delivery rates (Fig. 3), despite the skills of an experienced pilot with DGPS support.

- Rat eradication is practicable at seasons other than during late winter (i.e. July – September in the Southern Hemisphere);
- Circumstantial evidence from Curieuse indicates that rain-softened 20 ppm brodifacoum rodent pellets are eaten by cats, and that this bait is capable of incurring significant mortality;
- Private land tenure, human habitation and commercial tourism activities need not be viewed as barriers to alien mammal eradication projects. As in this case, island-based tourism activities can provide a sustainable means by which to restore and maintain threatened endemic biodiversity.

The current rodent-free status of Frégate Island would appear to be a case of good luck rather than good management. By September 2001, the rodent harbour fence, damaged by tidal action and poorly mended, was no longer rodent-proof; also, bait station maintenance and food disposal protocols were not being adequately implemented (Climo 2001). The recent re-invasions on Denis and Curieuse Islands illustrate the consequences of failing to implement rodent quarantine protocols – an impossible task unless there is total community awareness and support. The importance of acceptance and strict implementation of accepted rat quarantine protocols cannot be overemphasised (e.g., the landing of building materials, particularly roofing thatch on rat-free islands, without first fumigating the materials, poses an exceedingly high risk of rat invasion). Though simple in principle, this can be difficult to achieve in practice, especially if island owners are absent for long periods or there is a high turnover of itinerant island staff. Workers coming from Mahé, a highly-modified ecologically-degraded island, may have a limited appreciation of the importance of threatened endemic populations surviving on islands such as Frégate, their vulnerability to predation by rats and cats, how easy it is to accidentally bring rodents ashore with cargo, and how such carelessness can potentially affect them (i.e. ecotourism is an important generator of employment in the Seychelles). Hopefully, the level of conservation awareness among Seychellois will continue to improve as a result of BirdLife Seychelles' ongoing programme of community education.

The colonisation by rats of Frégate Island in 1995 was monitored and documented (BirdLife Seychelles reports 1995 - 2001; Jones and Merton 1995; Merton 1996, 1999; Thorsen *et al.* 2000). Never before had a rat invasion of a biologically important island been recorded in depth. This event provided the opportunity to test whether it was possible to eradicate a rat population during the colonisation phase, something rodent contingency protocols assume is feasible but had not yet been rigorously tested. Prompt intensive action while the invading population is localised and relatively small would, if successful, have minimised eradication costs and impacts on threatened species. Why then did it take five years to mount an effective rat eradication campaign?

The first rat was sighted on the island several weeks before conservation managers became aware of the invasion (Thorsen *et al.* 2000). Greater awareness among island owners and staff of the potential impacts of rats would have facilitated immediate reporting of the first sighting.

Although monitoring and control efforts were initiated promptly, the initial eradication attempt was seriously constrained by delays in obtaining funding and fears for the safety of threatened endemics at risk from rodenticide poisoning (Thorsen *et al*. 2000; Merton 2001). Clearly, as in the 1964 rat irruption on Big South Cape Island, some stakeholders, including local biologists, under-estimated the potential impacts of rats on the island's threatened endemics (i.e. the cost of doing nothing) and doubted the feasibility, and ecological and economic benefits of rat eradication (Gerlach, J. 1997, 1999; Gerlach, R. 2000).

In any such future event, funds may be more rapidly forthcoming if a conservation agency such as BirdLife International were to take a more pro-active role as watchdog and facilitator between private land owners and government agencies. Land owners, island staff, and all affected parties need to be made aware of the potentially-devastating impacts of rats on endemic island fauna, the feasibility of eradicating rats, of maintaining an island's rat-free status, and of where to quickly access expert advise and assistance.

The successful eradication of rats from Bird and Frégate Islands will, we hope, inspire similar effort on other islands with high biological values or potential – regardless of tenure or occupancy! However, the re-colonisation by rats of Denis and Curieuse Islands illustrates the need for ongoing conservation education and awareness programmes, which can ultimately foster a sense of responsibility, pride and stewardship among local communities. It can be difficult to assess the level of understanding of (and support for) a pest eradication proposal among the local population, particularly on islands where workers are often transient. However, the feasibility of eradication and quarantine projects on inhabited islands is more likely to be limited by lack of public support or awareness than by lack of technical capability.

Unfortunately, cats and rats are not the only introduced predators threatening Seychelles endemics. For example, barn owls (*Tyto alba*) and Indian mynahs are subject to control programmes on islands where they are predators of Seychelles magpie-robins, their chicks or eggs (Millett *et al.* 2000). The exotic yellow crazy ant (*Anoplolepis gracilipes*) has been present at low densities on Mahé since 1960 and has also colonised Denis and Bird Islands. The Bird Island population, first noticed in 1991, apparently remained small and localised until 1997, after rats were eradicated. By 1998, crazy ants had become widespread and abundant, with serious impacts on the island's endemic fauna and flora (Feare 1999). There is no real evidence that this ant irruption resulted from the removal of rats, as suggested by Feare (1999). Unfortunately, the re-colonisation by rats of Denis Island, where crazy ants remain at low density, prevents further evaluation of this theory. We agree that plans to eradicate alien vertebrates should include investigation of the presence of other exotic animals and plants that might benefit from the target species' removal. However, such "knock-on" effects are seldom easily predicted. Documentation of the major ecological impacts of the Bird Island crazy ant irruption should at least ensure that this species is not under-estimated in the future. Island quarantine protocols should extend to minimising the risk of importing crazy ants to new islands, monitoring to ensure their early detection, and implementing containment and eradication measures of any new colonies immediately they are detected.

ACKNOWLEDGMENTS

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Appendix 1 Summary of rat index trapping results, weights and measurements on four Seychelles islands ([1] prior to first poison application; [2] after first poison application; * Head and body length; [ ] number of traps set per night; TN trap-nights; CTN corrected trap-nights (see Cunningham and Moors 1996)).

	Trap nights	Rats caught	Rats/ 100TN	Rats/ 100 CTN	Weight adult male	Weight adult female	HBL* adult male	HBL* adult female
Rattus rattus								
Bird Is (Nov. 95)	108	22	20.3	26.5	132 70-180	139 55-190	175 145-190	168 115-190
Bird Is [1] (Nov. 96)	43 [23]	31	72	141	-	-	-	-
Curieuse Is (Jul. 98)	75	37	49	82.2	114 73-147	105 90-134	167 145-180	158 150-170
Curieuse Is [1] (Jun./Jul. 00)	109 [26]	61	56	84	114 61-205	101 75-150	167 145-180	160 135-180
Curieuse Is [2] (Jul. 00)	302 [26]	11	3.6	4	-	-	-	-
Denis Is (Jul. 98)	89	29	23.5	35.6	118 73-170	106 62-24	168 123-195	163 140-180
Denis Is [1] (May/Jun. 00)	49 [19]	14	28.6	35.4	119 70-147	112 94-133	163 155-174	153 146-160
Denis Is [2] (Jun. 00)	133 [19]	0	0	0	-	-	-	-
Rattus norvegicus								
Frégate Is (Jul. 98)	97	26	27	38.8	363 270-402	289 170-420	255 225-320	215 185-240
Frégate Is [1] (May/Jun. 00)	374 [25]	76	20.3	27.2	305 167-455	279 175-375	219 183-260	213 180-295
Frégate Is [2] (Jun. 00)	125 [25]	0	0	0	-	-	-	-

Appendix 2 Chronology of baiting and trapping events on four Seychelles islands

	Bird Is (101 ha)	**Denis Is** (143 ha)	**Frégate Is** (219 ha)	**Curieuse Is** (286 ha)
First application of rodent bait	30-31 Oct. 96 (Talon-stations)	2 Jun. 00 (pellets-aerial)	8 Jun. 00 (pellets-aerial)	2 Jul. 00 (pellets-aerial)
Helicopter flying time, excluding ferrying time (hours:minutes)	NA	2:30	2:15	2:45
Bait used (kg)	60	2375	3000	3700
Application rate (kg/ha) (heavier in coastal zone, lighter elsewhere)	0.6	16.6	13.8	12.9
Rainfall over subsequent three days	Nil	1.9 mm (Aride)	Nil	3.2 mm
First dead rat seen	3 Nov. 96	5 Jun. 00 (1)	12 Jun. 00 (6)	5 Jul. 00 (5)
Second bait application	8-10 Nov. 96 (pellets-hand)	11 Jun. 00 (pellets-aerial)	13 Jun. 00 (pellets-aerial)	13 Jul. 00 (pellets-aerial)
Helicopter flying time, excluding ferrying time (hours:minutes)	NA	1:05	1:30	2:00
Bait used (kg)	475	1000	2050	2900
Application rate (kg/ha) (heavier in coastal zone, lighter elsewhere)	4.7	7.0	9.3	10.1
Rainfall during subsequent three days	31 mm	0.9 mm	Nil	22.4 mm
Third bait application	18-20 Nov. 96 (pellets-hand)	-	7 Jul. 00 (pellets-aerial)	-
Helicopter flying time, excluding ferrying time (hours:minutes)	NA	-	1:40	-
Bait used (kg)	495	-	2625	-
Application rate (kg/ha) (heavier in coastal zone, lighter elsewhere)	4.9	-	11.9	-
Rainfall during subsequent three days	Trace	-	2.5 mm	-
Total pellet application rate (kg/ha) excluding block baits	9.6	23.6	35.0	23.0
Rat index trapping commenced	1 Nov. 96	30 May 00	24 May 00	25 Jun. 00
Rat index trapping ceased	17 Nov. 96	9 Jun. 00	13 Jun. 00	15 Jul. 00
Last rat trapped	13 Nov. 96	2 Jun. 00	8 Jun. 00	7 Jul. 00
Non-toxic cat baiting commenced	-	16 Jun. 00	-	17 Jul. 00
Toxic cat baiting commenced	-	23 Jun. 00	-	22 Jul. 00
Cat trapping commenced	-	26 Jun. 00	-	24 Jul. 00
First cat trapping session ended	-	11 Jul. 00	-	16 Aug. 00

Appendix 3 Rodent eradication and island quarantine – some urgent recommendations

Following is a summary of some urgent measures, which I regard as essential to the success of the rodent eradication project and to minimise the risk of re-invasion of Frégate Island by rats and mice.

Administrative responsibility and accountability

Assuming the current eradication attempt is successful, maintaining Frégate Island free of rats and mice will be difficult but by no means impossible. It will require a very high level of commitment and compliance by management and staff. A senior staff member must be given responsibility for overseeing island quarantine and ensuring implementation of a rodent quarantine and contingency plan. Duties of this person must include:

- ongoing promotion of rodent awareness within Frégate Island's management and staff, as well as the crews of visiting boats;
- urgent follow-up of any reported sightings of rats or mice on the island;
- regular replenishment of toxic baits in 86 permanent rodent bait-stations (on land and on boats);
- ordering supplies of bait as necessary;
- ensuring that stores and bulk cargoes arriving on Frégate are correctly packed, are opened inside a rodent-proof compound and are carefully checked for rats and mice.

Any sightings or suspected sightings of rats or mice on Frégate Island must be reported immediately to the Plantation Manager for urgent follow-up action.

Management and disposal of foods and kitchen refuse

It is essential that foods are inaccessible to rats and mice at all times.
All staff houses must have a mouse-proof cupboard or food storage area.
Waste foods and other kitchen refuse must be stored in sealed bins and disposed of in such a way that they are unavailable to rats and mice (i.e. fed to pigs, burned or buried).
Food scraps must never be thrown onto the ground.

Recommendations

- Establish an island rule making it an offence to dispose of food scraps and kitchen refuse other than in bins with tight fitting lids.

- Designated refuse collection sites must be established similar to those in parts of Mahé; ie a concrete platform with low walls, large enough to accommodate three wheelie-bins lined with plastic sacks - a bin each for burnables; food refuse for the pigs; and the third for items for shipment to Mahé for dumping.

- Refuse must be collected and disposed of daily - ie food refuse fed to pigs; burnables incinerated each day; and refuse for shipment off the island stored in large bins (skips) with tight-fitting hinged lids, to be sent off the island at the first opportunity.

- So far as is practical, catering should be centralised – storage and preparation of foods at staff houses should be discouraged.

Packaging and shipment of stores to the island

All stores must be sealed in rodent-proof containers before being transported from Mahé.
The "Coleman 150" polystyrene food containers currently used for cool-store items are ideal. Other foods must be packed in similar plastic or metal boxes with tight-fitting lids, which are to be closed and sealed immediately after packing. Cardboard cartons are likely to harbour rats and mice and must not be used.

All bulk cargoes and containers (especially thatch and building materials) must be fumigated in Victoria. Doors must be locked immediately after, and not opened until arrival on the island.

Appendix 3 continued

Unpacking of stores and bulk cargoes arriving on the island

A rodent-proof compound in which to unpack all stores and equipment is essential and urgently needed to prevent re-infestation of the island by rodents. This rat and mouse-proof compound - either a large storeroom or a rodent-fenced area - must be large enough to take a trailer loaded with a shipping container. On arrival on the island all stores and bulk items (including containers) must be taken immediately to this compound and the gate/door sealed during unpacking. Any rodent can then be confined and destroyed.

Boats visiting the island.

The rodent-proof harbour fence must be completed and maintained to a high standard as a matter of great urgency.

All vessels calling at Frégate must use the boat harbour. Beaches or surge basins must not be used for landing since these sites are not protected by a rodent-proof fence.

All vessels that visit Frégate regularly or occasionally must have at least one bait station loaded with rodenticide bait permanently on board. Baits inside these stations must be checked and if necessary replaced by the island rodent officer each time the vessel visits the island.

Maintenance of permanent rat bait stations

The 86 permanent bait stations positioned near landing sites and other potential rodent “hot-spots” on the island must be serviced regularly if they are to be effective. The rodent officer must ensure that rodenticide baits in each station are carefully checked each month (or more regularly if required) and that baits are replaced as necessary.

Reducing cover for rodents

Rodents thrive in dense cover and cannot survive without cover in which to hide by day. In spite of commendable recent efforts, Frégate Island offers an abundance of cover for rodents. High, ongoing priority must be given to reducing cover - and thus potential rodent habitat.

Recommendations

Higher priority must be given to the clean-up process - removal from the island of all refuse and discarded materials. The long-abandoned African tent camp at Plaine Magnan is a case in point.
Disposal of slashed vegetation and fallen coconut fronds is a problem on the island. Rather than accumulating this material at a few traditional sites (and so creating substantial areas of prime cover for rodents), it may be practical to mulch or disperse some in forested areas, so reducing the pressure on traditional dumping sites.

Don Merton
26/06/00

Eradication of rats and rabbits from Saint-Paul Island, French Southern Territories

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Abstract Introduced black rats (*Rattus rattus*) have decimated the seabird colonies on Saint-Paul Island (Southern Indian Ocean). Only six of the 13 seabird species originally breeding on Saint-Paul are now represented by only a few individuals confined to an islet located 150 m from the main island. This led us to believe that recolonisation was possible on Saint-Paul Island if all rats were removed from it. The Administration of Terres Australes et Antarctiques Françaises decided to eradicate rats and part of the funding was provided by the European Development Fund. Two preliminary trials were conducted in 1995 and 1996, and in January 1997 13.5 tonnes of brodifacoum bait (Pestoff Rodent Bait) were spread by helicopter. The island was intensively checked for rat presence during three months after the drop and during two more follow-up operations in late 1997 and early 1999, when respectively 48, 18 and five rabbits (*Oryctolagus cuniculus*) were killed. We are now confident that black rats are eradicated but eradication of rabbits still needs to be confirmed. Mice were not eradicated, presumably due to lack of good cover of baits, linked to spreader malfunction. The Saint-Paul Island project demonstrates the efficiency of the aerial technique against rats, but shows that rabbit eradication needs a more sustained effort. Breeding of endemic Macgillivray's prion (*Pachyptila macgillivrayi*) and of great winged petrels (*Pterodroma macroptera*) has already begun on Saint-Paul Island.

Keywords *Rattus rattus*; rabbit; introduced species; eradication; *Pachyptila* sp.

INTRODUCTION

Saint-Paul Island (38°42"30'S, 77°32"30'E) belongs to the French Southern Territories whose management is regulated by the Administration of Terres Australes et Antarctiques Françaises (TAAF); together with Amsterdam Island, 80 km to the north, they are among the most isolated islands in the world, being 3200 km from Australia, 4200 km from South Africa and 3300 km from Antarctica (Fig. 1).

Amsterdam Island and Saint-Paul Island offer a classic example of seabird decline after introduction of alien mammals. There are several aspects of environmental protection by TAAF, and restoration programmes are the most recent (Jouventin and Micol 1995). In 1988-1989 a programme of rehabilitation was carried out on Amsterdam Island (55 km^2; Micol and Jouventin 1995; Micol *et al*. 1999), with the island subdivided by an 8 km long fence and the removal of feral cattle from one side. This allowed the protection of Amsterdam albatrosses (*Diomedea amsterdamensis*) and of native vegetation. Cats (*Felis catus*) and rats (*Rattus norvegicus*) were still a threat to smaller birds and Amsterdam Island was too large to attempt control of these pests. Saint-Paul Island, 80 km south, is a smaller island of 8 km^2, consisting of the eroded top of a single volcano rising to 268 metres. It is now ear-shaped as on the lower east side, the rim of the crater has broken down and been invaded by the sea (Fig. 1). Since time immemorial, this vast sheltered amphitheatre of bare basaltic rocks has been a favoured spot for fur seals (*Arctocephalus tropicalis*) and elephants seals (*Mirounga*

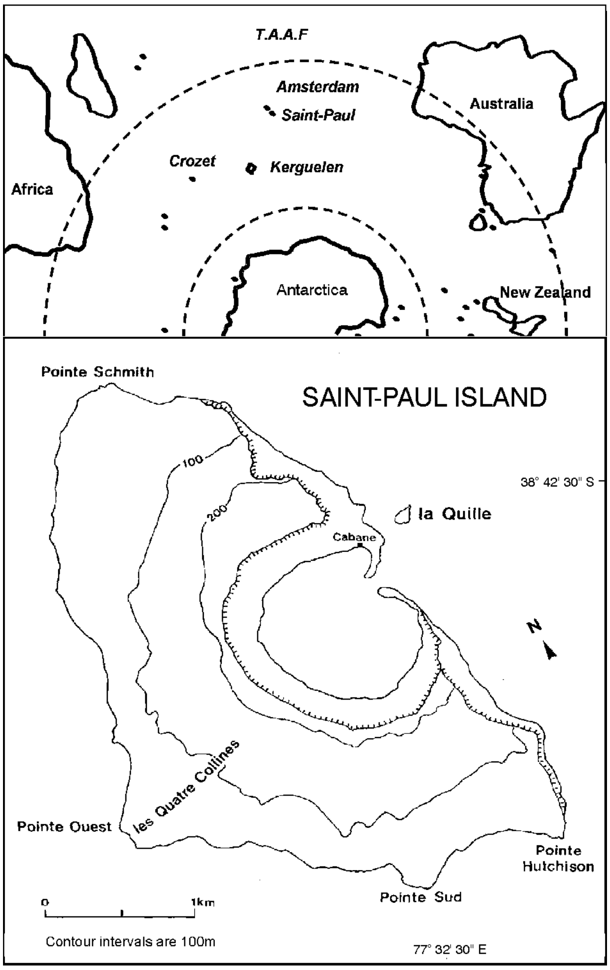

Fig. 1 The location (upper) and features (lower) of Saint-Paul Island.

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leonina) to breed and bear their young. Apart from the inner shores of this shallow crater, the terrain is very broken and hard to traverse. Dissected rock flows descend sharply from the high crater rim down to the sea to end abruptly in precipitous cliffs. There are some springs of hot water and other signs of active volcanic activity smouldering beneath the surface. Smoke or steam sometimes issues from the dark basalt walls surrounding the basin. Saint-Paul Island lies too far north to be within the true subantarctic zone, but has a relatively mild, wet oceanic climate like that of Tasmania and New Zealand at the same latitude. Mean sea-level temperature is 13.8°C. The coldest month is August with a mean of 11.2°C, while the warmest is February with a mean of 17.0°C. Mean annual rainfall is 1115 mm with a short dry season in summer (February-March) when evaporation exceeds rainfall (data from the Meteorological Office Recording Station on Amsterdam Island).

Saint-Paul Island was discovered in 1559 and subsequent sightings of the island occurred throughout the seventeenth century, but the first detailed description of it, and perhaps the first landing, was by William De Vlamingh in 1696 (Richards 1984). From its discovery, the island has never been permanently inhabited. While on these islands the transient sealers almost exterminated the subantarctic fur seals between 1790 and 1810, they also decimated the original flora and fauna through repeated carelessness with fire and through the introduction of new species (Richards 1984). Through accounts of sealers (Peron 1824), fishermen, sailors, shipwrecked mariners and scientific expeditions (Velain 1878), it is possible to provide a general description of the early flora and fauna of the island. The shore was covered with such a multitude of seals that visitors were obliged to disperse them before they could land. The whole island was covered with a kind of coarse long grass or reeds and yielded various seabirds, especially a blue petrel, as hundreds of thousands of breeding pairs were nesting in the rocks.

As with most of the subantarctic islands visited in pursuit of fur seal skins (Johnstone 1985), many of these visits lasted several years, and the men frequently, but unintentionally, brought rats and mice with them. Cats were subsequently introduced for controlling rodents. Pigs (*Sus scrofa*), goats (*Capra hircus*) and rabbits (*Oryctolagus cuniculus*) were also introduced as food. All of these introductions contributed to the destruction of the seabird populations. Before the start of this project only rabbits, rats (*Rattus rattus*), and mice (*Mus musculus*) still existed on Saint-Paul Island. Rats and rabbits have a negative impact by preventing recolonisation of seabirds, the first preying on the birds, the second using the same burrows. From subfossil bones found on Amsterdam Island (Worthy and Jouventin 1999), we know that several species of petrels and prions are now extinct from the Amsterdam/ Saint-Paul Islands group. However, some seabird species now extinct on Saint-Paul Island and Amsterdam Island breed on nearby La Quille islet, 150 m from Saint-Paul Island, which has no introduced mammals, but is only 1 ha in size (Fig. 1).

This paper describes the rat eradication on Saint-Paul Island, which comprised four phases: (1) feasibility study in New Zealand in 1994, (2) two exploratory surveys in 1995 and 1996, (3) the eradication campaign in 1997, and (4) two follow-up surveys in 1997 and 1998-99.

The Administration of TAAF funded phases 1 to 4 and the European Development Fund co-funded phases 1 to 3.

METHODS

Phase 1 - Feasibility study in NZ

The feasibility of the operation, and the best method to use had to be assessed before starting the project. In 1994 a feasibility study was conducted in New Zealand by one of us (TM), consulting with people involved in similar operations from Department of Conservation, Auckland Conservancy (David Towns and Ian MacFadden) and from Landcare Research, Nelson (Rowley Taylor and Bruce Thomas). A comparison was established between the two most widespread techniques; aerial drop versus bait stations (Taylor and Thomas 1989; McFadden and Greene 1994). The first was more rapid, and the second was more likely to prevent primary poisoning of non-target species. It thus appeared that the eradication was feasible and that, with the presence of numerous cliffs around Saint-Paul Island and the absence of non-target species except non-breeding skuas, the use of an aerial drop was the best solution. We decided to focus on the eradication of rats as they caused the most significant damage to the avifauna. Eradication of rabbits was also planned as possible but not certain, as it was the first time these two species were involved together in an eradication programme. A previous eradication of rabbits on an island of similar size, Enderby Island (700 ha), had been successfully conducted in 1993 (Torr 2002). It appeared that 99% of the rabbit population were killed by poison but that the remaining rabbits had to be killed with dogs and guns. We then assumed that the same schedule would be applicable to Saint-Paul Island and that for a successful eradication we would need hunting experts and dogs.

Phase 2 - Exploratory surveys

Two exploratory surveys were conducted in February-March 1995 (four people) and April-July 1996 (five people). The aim of these surveys was to estimate (1) the density and the dispersal of target species as some data on the biology of rats, (2) the palatability of baits and (3) the status of seabird populations.

Target species survey

Rats. Densities of rat populations were assessed using snap traps. Forty sites were chosen around the island, 15 along external coasts, six at mid altitude of the external slopes, nine at the edge of the inside crater, and 10 along the coast of the inside crater. At each site, a line was established, each 100 m long and consisting of 10 traps spaced 10 m apart. The trap-lines ran parallel to the shoreline when applicable. Each site was sampled once and the sampling consisted of 2-3 nights during which traps were checked once or twice. In addition to the snap traps, some live traps allowing multiple captures were set occasionally near the penguin colony and near the flesh footed shearwater colony.

After having determined the distribution of rats we then wanted to assess what would happened if by mistake an area was not poisoned. Do the rats living in an area with no poisoned baits move into a nearby poisoned area where rats have disappeared?

Ten snap traps were set around the hut in order to create a sink experiment simulating the death of rats in a poisoned area. At the same time we used radio telemetry to investigate sizes and spatial distribution of home ranges of 12 rats in an area located 500 m away. Adults were fitted with radio transmitters in the field under anaesthesia and released upon recovering. Radio collars consisted of button cell tags (Biotrack Ltd, Wareham, UK) that were mounted on plastic cable ties and coated with acrylic. Transmitters weighed on average 6.5g, 3% of the mass of the average 200 g rat to which they were attached. We located rats using two three-element Yagi antennas. Rats were tracked regularly at night from 10 May to 11 June 1996.

Rabbits. Rabbit populations were assessed using visual counts of faeces and observations of living rabbits along transects across the island while we were in search of seabirds and rat presence.

Palatability tests

Bait palatability was assessed with 100 kg of non-toxic baits, some coloured with the dye Rhodamine B which colours the internal tissues and faeces of consumers pink. To determine the proportion of rats and rabbits having eaten baits, dyed baits were spread by hand over a 1.5 ha area. In order to test possible shyness due to Rhodamine B we did the same experiment in an adjacent area with non-dyed baits. After spreading the pellets we marked in each area five 100m² control sites where: (1) we counted the number of baits on the ground, and (2) removed all rats' and rabbits' faeces. We then checked the number of baits remaining per site daily, and we counted the number of faeces, coloured or not, before removal. A first trial at 10 kg/ha was conducted in a high rat density area (HRDA) and in a low rat density area (LRDA). A second trial at 20kg/ha was conducted in the HRDA.

Estimation of seabird populations

Saint-Paul Island was known to support antarctic terns (*Sterna vittata*), sooty terns (*Sterna fuscata)*, flesh footed shearwaters (*Puffinus carneipes)*, rockhopper penguins *(Eudyptes chrysocome)* and sooty albatrosses (*Phoebetria fusca)* (Segonzac 1972). Some large and dark petrels were seen flying in the vicinity of Saint-Paul Island (Segonzac 1988) but the exact species and the status are unknown.

An intensive search for seabirds was carried out in 1995 on Saint-Paul Island by night sightings, visual inspections of burrows along the coasts and listening for songs at night. New species were particularly investigated, as it was the first such survey for a long time.

Phase 3 - Eradication

Following the feasibility study, the rat breeding biology was not important for the timing of the operation and suitable weather conditions was the priority as the bait breaks down in the rain. The dry season of January-February was chosen as the best time. Two drops of 10 and 5 kg/ha separated by 3-4 weeks were planned but the calendar of the ship supplying TAAF did not allow this and we had to manage with only one drop.

The bait was spread using a Lama helicopter (HéliRéunion, France) with a bait bucket (like a monsoon bucket with a base plate adapted to spread the bait and regulate the bait flow). Allowing for the speed and height of helicopter flight necessary to get a good coverage of baits, the whole coverage of the island by the helicopter would last 10 hours. Fifteen tonnes of Pestoff Rodent Bait (a 2 g grain based pellet containing brodifacoum at 20 ppm), and the bucket, were ordered from Animal Control Products (Wanganui, New Zealand). For safety reasons, the helicopter was due to fly with the supply ship Marion-Dufresne around Saint-Paul Island. The ship was planned to stay six days at the island, in order to get at least two days of good weather conditions. The bucket was loaded with baits directly from two sites on the island, with staff moving to the second site after completion of the first site. Six people were necessary for refilling the bucket, two opening the bags, two pouring the baits in the bucket, and two holding the empty bags.

Following the airdrop approximately 300 kg of Pestoff Rodent Bait was spread by hand in some places missed by the drop.

Coverage check

After departure of the ship, five people and two dogs stayed for three months on Saint-Paul Island to check the coverage of baits and to search for sign of rats and rabbits. During this stay there were also searches for any signs of non-target losses.

From the 8 February 1997 a total of 260 stakes to check for rat sign were set 100m apart along the external coast, the edge of the crater, the internal coast of the crater and along eight lines running from the top of the crater to the coast (Fig. 2). Candles and slices of sausage were placed on each stake and checked biweekly. Additionally, 100 snap traps baited with sausages, fish or apples were set near these stakes and moved each three days to check new sites.

Phase 4 – Follow-up surveys

A follow-up using methods similar to those used in 1997 was conducted in November-December 1997, with six people and the two dogs used during the eradication campaign.

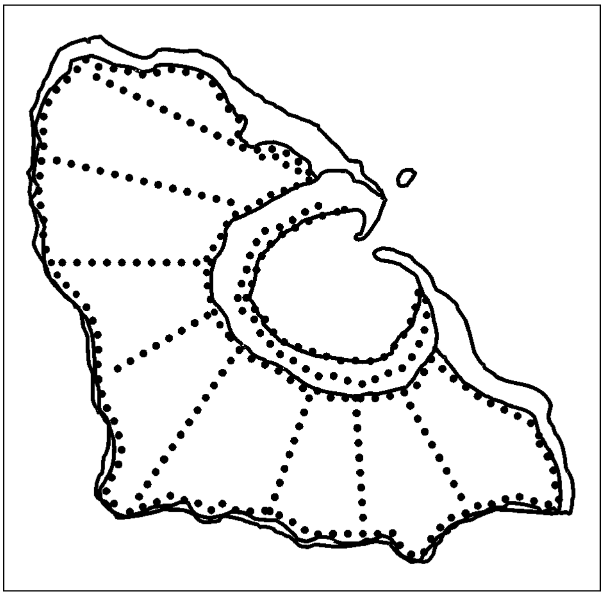

Fig. 2 Location of post-drop monitoring sites. Each dot represents a stake set 100m apart.

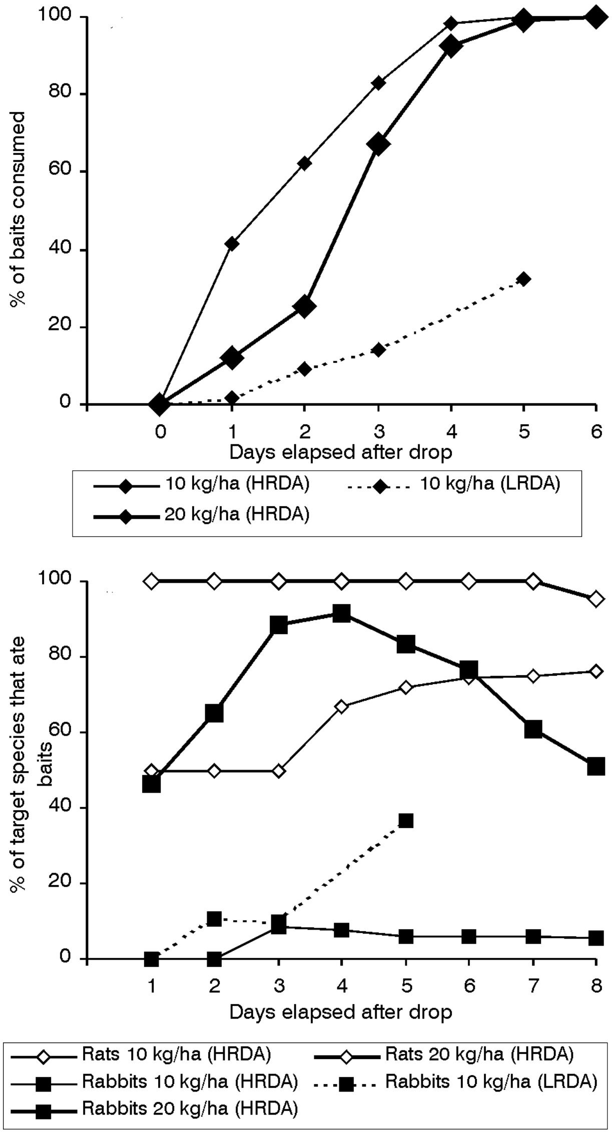

Fig. 3 Upper: Bait-take in non-toxic trials. Lower: Target species consumption of bait during non-toxic trials.

Unfortunately one dog died before arriving on Saint-Paul Island and the job had to be done with only one dog. A last follow-up was organised in December 1998–February 1999 with five people and two dogs.

RESULTS

Exploratory surveys

Density of target species

Signs of presence of rats and rabbits were found all around the island, with the highest density inside the crater.

The penguin colony live trap was very successful and on one occasion caught 12 rats together. Overall trapping results indicated that rats were 10-20 times more dense in the crater than the outside part of the island, varying from 5/ha on most of the island to 100/ha around the penguin colony and the hut. Rat density seemed to be determined mainly by food, as they were more numerous in and around the penguin colony, and around the hut where the vegetation was denser.

Bait palatability

One hundred per cent of the baits of both 10kg/ha and 20kg/ha were eaten four days after their dispersal in the HRDA but only 30% were eaten in the LRDA five days after its placement at 10 kg/ha when rain caused desegregation of the pellets (Fig. 3). The second trial conducted with baits at 20 kg/ha in the HRDA showed that ninety percent of the baits had disappeared five days after the dispersal.

When checking the HRDA spread at 10kg/ha, a maximum of 76% of rats and 8% of rabbits had eaten baits eight days after bait dispersal (Fig. 3). The trial at 20kg/ha in the same area gave better results with 100% of rats and 92% of rabbits having eaten baits four days after the spread.

It was thought that rats consumed baits before the rabbits, which may have left no available bait to eat. The relative density of baits and target species was thus very important when trying to eradicate the two species together.

The palatability tests confirmed that rat eradication on Saint-Paul Island was achievable using the Pestoff Rodent Bait but that not all the rabbits would be killed, as was the case on Enderby Island (Torr 2002).

Movements of rats

At the start of the radio tracking study in the penguin colony, rats moved no further than 100 m. A total of 167 rats were then killed from 28 April to 26 June 1996 around the hut. After creating this ‘sink’ around the hut, at least one rat moved away from the penguin colony and was caught twice in a live trap around the hut before returning to the penguin colony 500 m away. It was thus believed that even if an area was missed with baits we could obtain an effective eradication of rats.

During the same time three radio-tracked rats disappeared, being eaten by skuas. Based on the New Zealand experience, most poisoned rats were expected to die in their burrows, thereby reducing the risk of secondary poisoning of the 10-12 non-breeding skuas living in the area.

Seabird species

In 1995 the only small petrels to breed on Saint-Paul Island were confirmed to be 20-100 pairs of storm petrels in a 60m elevated cliff located south of the island. Ten to fifty pairs of white-bellied storm petrel (*Fregetta grallaria*) were discovered and also 10-50 pairs of Wilson's storm petrel (*Oceanites oceanicus*) that were not previously known to breed in this area. It was also confirmed that there were still five petrel species breeding on La Quille: endemic Macgillivray's prion (*Pachyptila macgillivrayi*); fairy prion (*Pachyptila turtur*); little shearwater (*Puffinus assimilis*); white-bellied storm petrel and we established that 40-60 pairs of great-winged petrels (*Pterodroma macroptera*) were breeding on La Quille.

Eradication

Poisoning

From the results of the trials it was calculated that 15 tonnes of bait were needed. The bait was shipped from New Zealand through Singapore taking two months to reach La Réunion. Some bags arrived mouldy, as there was a problem with condensation from the ceiling of the container dripping over the product. Each of the six hundred 25kg bags was thus opened to sort out the bad pellets leaving 13.5 tonnes available for the drop. It was decided to proceed. The condensation problem was solved by erecting a polythene tent inside the container for later shipments and it seems that no more trouble has occurred with bait going mouldy.

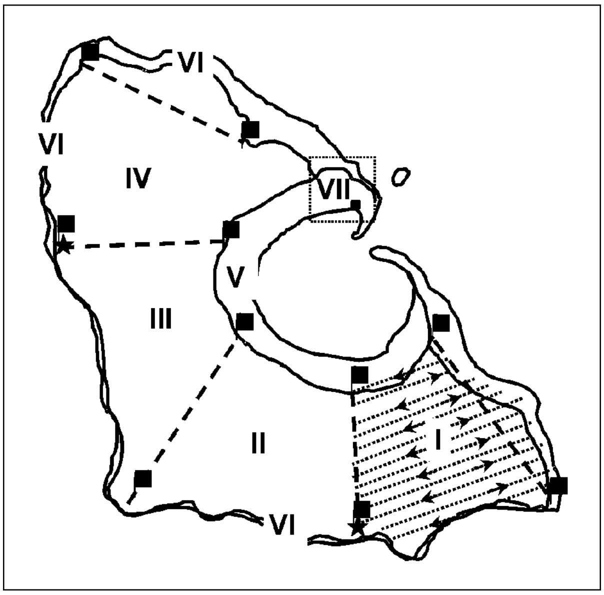

Fig. 4 Bait treatment areas and flight lines. Areas I-IV = low density 10kg/ha, area V = high density 20 kg/ha, area VI = low density 10kg/ha and area VII = very high density up to 40kg/ha. Stars indicate the sites where the bucket was loaded with baits. Flags represent the extremities of the lines with people indicating the transect lines to the helicopter. Dotted lines represent those flight lines.

The island was divided into seven main areas to be covered with bait densities varying from 10 kg/ha to 40kg/ha (Fig. 4). Poison was dropped by helicopter, following parallel lines as indicated by two groups of four people moving with flags along lines running from the top of the crater to the coastal cliffs. Four main areas were determined that covered the outside part of the crater where the target species densities were the lowest. A fifth area covered the inside cliffs of the crater and was covered at 20 kg/ha, and a sixth area covered the outside cliffs where baits were laid directly from the bucket at 10 kg/ha. The seventh area covered the penguin colony and surrounds and the hut where baits were spread up to 40 kg/ha.

For each area, in order to have the best coverage by the helicopter, transect lines spaced 100m apart were marked at each end by people with flags, each person moving to the next position after having been overflown by the helicopter. Two teams of three people were initially tasked to mark the lines but once the drop started it appeared that the helicopter crossings were faster than people could run over the tussocks and one more person was dedicated to each line. Thus eight people were needed for showing tracks to the helicopter and six to reload the bucket. Each group had a VHF radio and a supervisor coordinated the groups, the helicopter, and the ship. The airdrop of 13.5 tonnes of Pestoff Rodent Bait was conducted on the day of arrival on Saint-Paul Island, 21 January 1997. The forecast was good for the following days and no rain or heavy wind occurred during the operation.

The total aerial operation was due to last 10 hours but after a few hours work the engine on the spreader-bucket malfunctioned. The spreading was finished without the spinner (i.e. with baits falling directly from the bucket). In order to increase the coverage width without the spinner, the helicopter pilot moved the helicopter from side to side. This meant that the dropping operation lasted longer than planned and was finally finished on the morning of 23 January after a day off because the ship was not operational.

Coverage check

Dropped pellets missed only one area which was consequently hand-baited with 100 kg Pestoff Rodent Bait. The areas covered when the bait spinner was out of order were alternatively covered and not covered with baits, along bands approximately 50m wide. This was estimated to be enough to achieve the rat eradication.

At the conclusion of the eradication follow-up in April 1997, 12 dry bait stations were left in particular areas as a precaution against any remaining rats.

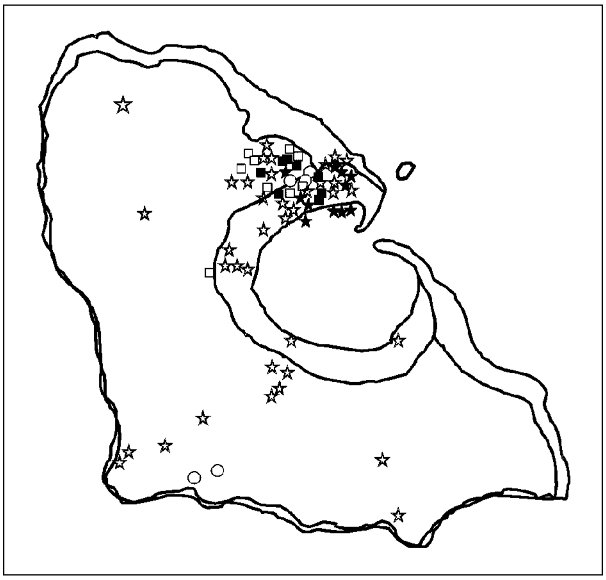

Fig. 5 Rabbits killed using dogs and guns during the follow-up surveys, from 10 Feb to 7 April 1997 (white star adult, black star young) and 25 Nov. to 18 Dec. 1997 (white square adult, black square young) and 25 Dec. 1998 to 18 Feb. 1999 (white circle adult).

Sign of rats and rabbits

The first dead rat was found five days after the bait drop and the first rabbit seven days after the drop.

The last living rat was seen two weeks after the bait drop.

Cockroaches around the hot springs ate some candles and some sausage slices from the stakes that were set out to signal any rat presence. Although mice nearly disappeared during the first weeks, they reappeared in late March 1997 and also ate baits, and were sometimes killed by snap traps. During the three-month stay in 1997, and during the follow-up surveys, no rat sign was found although the island was covered tens of times, while checking of the stakes and while looking for rabbits.

From 10 February to 8 April 1997 we killed 48 rabbits of which 17 had eaten baits (Fig. 5) as revealed by stomach inspection. Some rabbits known to live in areas where baits were hand spread 4-5 times never ate baits and were only killed after a long period of hunting. From November to December 1997, we killed 18 rabbits and from December to February 1999 we killed 5 more (Fig. 5). Most of the surviving rabbits were killed in areas vegetated with rushes (*Juncus effusus*).

Non-target species

Not a single skua was found dead from January to April 1997 and their numbers were the same as before the operation (10-12 individuals).

Mice

The last mouse was trapped on the 11 February and we did not see a living mouse before the end of March 1997.

DISCUSSION

Eradication of rats and rabbits

The eradication did not proceed according to our original plan but we were able to proceed with a revised plan, and there was no rat activity at the stake stations at the end of the main campaign in April 1997. However, the bucket malfunction illustrates the need for testing all equipment before it is sent to remote places such as Saint-Paul Island.

The absence of any sign of rats on Saint-Paul Island during checks made for three months then 11 months and 24 months after the airdrop confirms the effectiveness of the method.

The good weather conditions with no rain for two months after the drop was a very important factor, as baits remained available and viable all during this time.

After more than two years elapsed, and three surveys, it is now certain that eradication of rats has been achieved. However, as we have had no time after the last rabbit was killed to check the complete island one more time, it is too early to be confident that they are all gone. A last survey will be carried out at Saint-Paul Island in November-December 2001.

Eradication of mice

The baits killed a large percentage of the mouse population but we believe that the presence of non-baited areas due to spreader malfunction allowed some mice to survive and to recolonise the island. We also believe that the inability to do two bait drops prevented the effective eradication of mice. Mice reappeared more numerous than they were, presumably because of the removal of rat predation.

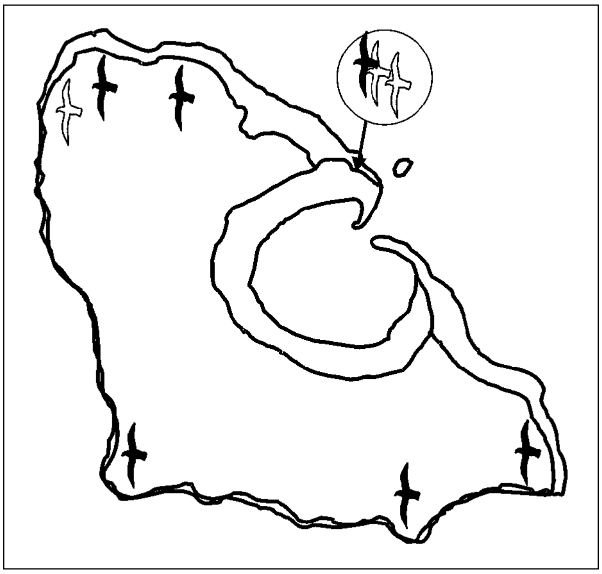

Fig. 6 Location of seabirds found after the eradication. Great-winged petrel (black symbols), Macgillivray's prion (white symbols).

It appeared that mice and cockroaches ate baits left in bait stations, making it doubtful whether the bait stations would have killed any surviving rats.

Recolonisation by seabirds

Two great-winged petrels were found in the flesh-footed shearwater colony on 1 March 1997. Prior to our departure the 9 April 1997, five other places were also found to support great winged petrel individuals (Fig. 6). It is believed that petrels did not re-establish on Saint-Paul Island because of the rat and rabbit activity. Petrels are extremely philopatric birds and they always breed in the same place, young usually being recruited to their natal colony. Because breeding places were limited on La Roche Quille, birds tried to breed on Saint-Paul Island, but they were disturbed or preyed on and no successful breeding occurred.

The first sign of endemic Macgillivray's prion on Saint-Paul Island was found on 11 February 1999 when we discovered three burrows with feathers and a strong smell under rocks located in front of La Roche Quille. Subsequent searches indicated that there were 5-10 pairs frequenting burrows in nearby areas. One lost egg was found intact in one of the burrows. In the same area 10-12 burrows were still being used in February 2000 (D. Pinaud, pers. comm.).

Modelling of the recolonisation processes by Macgillivray's prions suggests that populations will not recover for decades, but preliminary results are very promising.

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Cat eradication and the restoration of endangered iguanas (*Cyclura carinata*) on Long Cay, Caicos Bank, Turks and Caicos Islands, British West Indies

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Abstract Endangered Turks and Caicos rock iguanas (*Cyclura carinata*) are being displaced on Big Ambergris Cay by an expansive development project. We chose Long Cay, Caicos Bank, as a relocation site for some iguanas because it: (1) is a large (111 ha), uninhabited, protected reserve, (2) previously supported iguanas but did not have a current population, (3) could support thousands of iguanas, and (4) had no native mammals, few scavenging birds, and no nesting colonies of scavengers. There was a small population of feral cats, well-known iguana predators. To restore the island, we conducted an intensive cat poisoning campaign using sodium monofluoroacetate (1080), in July 1999. In November 1999, a test-group of 25 iguanas was taken from Big Ambergris Cay to Long Cay. Since their successful establishment we have relocated more than 400. The first hatchlings were confirmed in January 2001. Occasional trapping may be necessary to maintain Long Cay free of cats. We have begun patrols and courtesy visits to vessels cruising the area, installed informative and cautionary signs, and produced public service announcements for TV to reinforce the importance of keeping domestic animals away from uninhabited islands.

Keywords Feral cats, *Felis catus*; eradication; iguanas, *Cyclura carinata.*

INTRODUCTION

"When you have hope… that's what you've got."
Tom Sinclair

Biologists who work with rare species and their sustaining resources quickly realise that there is rarely good news to report. Ecological milestones typically range from "Good News - We Found One," to "Good News - They'll Only Be Destroying 90% of the Place!" We measure success on a relative scale, somewhat like a bug being rinsed down a sink drain who manages to hook his foot on the lip of the pipe and pulls himself out of the vortex… temporarily.

Considering all this, it is delightful to have any opportunity to report an ecological improvement: "Good News - We Have Successfully Restored an Island and Have Repopulated It With Endangered Turks and Caicos Rock Iguanas (*Cyclura carinata*)". In order to repopulate, we eradicated a population of predatory feral cats that was responsible for extirpating the native iguanas there. The habitat was then suitable and available for iguana re-colonisation. As of November 2000 we had restored a population of 400 iguanas to the island. They have been thriving and now have offspring. Perhaps this case history can serve as a template for those who are still hopeful.

Background

Human expansion, development, and the biotic baggage that arrives with them have adversely affected most Caribbean iguanas. Impacted taxa include the Lesser Antillean iguana (*Iguana delicatissima*) and all eight species in the genus *Cyclura*. Human predation, habitat reduction through clearing, burning, farming, and building, introduction of exotic competitors for food (feral ungulates), as well as exotic predators (dogs *Canis familiaris*, cats *Felis catus*, and mongooses *Herpestes auropunctatus*) have, on many islands, completely extirpated populations of endemic iguanas.

For example, in the British Virgin Islands where three of us have worked, no one remembers normal populations of the Anegada rock iguana (*Cyclura pinguis*). This species, once widespread on the Puerto Rico Bank, has slowly been reduced, by a number of human-related factors, to one naturally occurring population of fewer than 200 individuals (Mitchell 1999a, 1999b). The most critical problem for adult Anegada iguanas is livestock, which out-compete them for food. Adults are large and, although dogs can and do kill them, they are too big to be taken by cats. Cats kill and eat smaller hatchlings and subadults. Cat predation is the newest threat on the island, because of a town dump, established in the 1990s, that subsidised rapid growth of a feral cat population (Veitch 1998). Anegada needs habitat preservation and restoration, and radical management of exotics if iguanas are to persist there (Veitch 1998, Mitchell 1999a, 1999b, 2000). As of the year 2000, there has been no movement to conserve or restore the island, consequently the relocation of Anegada iguanas to two other islands with exotic species control, Guana and Necker, has proved prudent, successful, and may have temporarily saved *C. pinguis* from extinction (Goodyear and Lazell 1994; Mitchell 2000).

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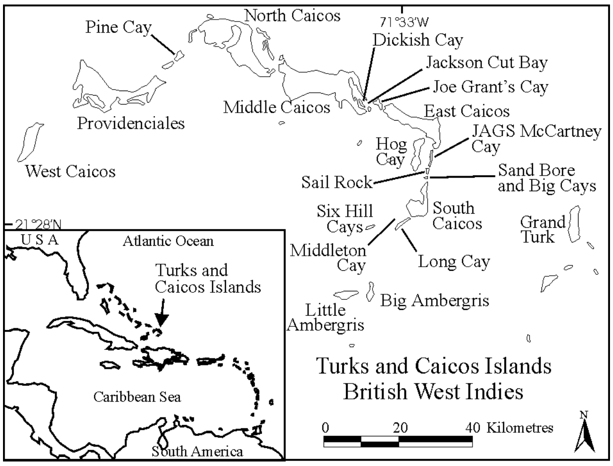

Fig. 1 The Turks and Caicos Islands.

A similar problem is becoming critical in the Turks and Caicos Islands (Fig. 1). Unlike Anegada iguanas, Turks and Caicos rock iguanas (*Cyclura carinata*) are still numerous on some islands. Gerber and Iverson (1999) estimated the total Turks and Caicos population at about 30,000 individuals. Normal densities in undisturbed populations approach 30 adults per hectare (Iverson 1979). As impressive as these numbers may sound, entire island populations are proving to be extremely vulnerable. Iguanas have completely disappeared from 13 Turks and Caicos islands over the past 20 years. Since most of the islands are relatively large, the range of the species was reduced from 500 km^2 to 28 km^2 (or 6% of its former extent) during this period (Gerber and Iverson 1999). In the 1970s John Iverson determined the principal reason for the decline. Between 1974 and 1976 he studied a population of 5000 adult *C. carinata* on Pine Cay, Caicos Bank, and documented its decline as a hotel was constructed there. By 1978 he could find no evidence of iguanas on Pine Cay although they were not completely extirpated (Smith 1992). Iverson linked the drop in numbers to free-ranging pets (cats and dogs) owned by the new hotel staff and residents (Iverson 1978). Adult *C. carinata* are much smaller than adult *C. pinguis*; consequently, cats prey on all age groups. In the Turks and Caicos Islands cat predation has proved the most serious problem for iguanas.

Currently, the two largest remaining populations of *C. carinata* occur on two remote islands on the Caicos Bank: Big and Little Ambergris Cay. Both are more than 30 km from the nearest inhabited cay. As of 1998 each island was thought to support populations of 15,000 adults (Gerber 1998). Commendably, the National Trust of the Turks and Caicos Islands has entered into a 99-year lease to protect Little Ambergris. Big Ambergris, however, will probably be developed. The developer's planned build-out leaves only small areas of the cay undeveloped and most of the iguanas there will be displaced.

As a mitigative measure, the Government is requiring the relocation of some of these endangered iguanas. In 1998 three agencies (The Department of Environment and Coastal Resources of the Turks and Caicos Islands (DECR), British West Indies; The Conservation Agency; and The Denver Zoological Foundation, the latter two non-governmental organisations from the United States) joined in a collaborative effort and began to discuss strategies for moving some of the animals. The problem was where to put them.

Based on prior experience we were confident that a relocation effort would be successful if we identified a suitable site (Goodyear and Lazell 1994; Knapp 2000). During our visit to the Turks and Caicos we examined potential targets for iguana translocation. Initially we searched for cays with the following characteristics: (1) no current population of iguanas, (2) no feral mammals (cats, dogs, donkeys, cattle, goats) that either prey on iguanas, compete with them for food (Mitchell 1999), or trample their burrows, and (3) suitable habitat (a diversity of food plants, loose sand for nesting, rocky retreats). To avoid mixing what might represent different genetic stocks we tried to locate translocation sites on the eastern Caicos Bank as near as possible to Big Ambergris Cay. Large islands were also considered desirable because they would accept a larger population of displaced Big Ambergris animals. A larger translocated population also would be less vulnerable to stochastic or catastrophic events that might cause local extinction.

We decided it was not prudent to introduce Big Ambergris iguanas to an island with a pre-existing iguana population. Because the iguana is fecund, we assumed that all islands currently supporting iguana populations would be at carrying capacity under the existing environmental conditions on each. New immigrants would cause stress to both groups of animals and presumably the population density would return (through mortality) to the original level and there would have been no net benefit from the relocation. We therefore opted to select among islands without iguanas.

METHODS

Relocation site selection

Potential targets for iguana translocation were examined in January 1999. Working from a 1995 report to the National Trust by Glenn Gerber (1998), as well as recommendations from islanders familiar with the uninhabited cays, we identified a number of islands to survey. Using a boat supplied by the DECR, we examined 15 islands. Six of these cays were known to support iguana populations; from these we developed a baseline for habitat quality and relative population status.

We verified iguana presence using one or several of the following indicators: tail drags in soft substrates such as sand or mud; dung; burrows with tail drags; or actual iguana sightings. Qualitative judgments of relative abundance were made from the number of sightings or signs. Small islands were surveyed completely by walking transects; on large islands we attempted to sample as much potential habitat as possible. We searched for areas with sand or soft soil that might accommodate iguana burrows or nests. On each island we noted the composition of the plant community, particularly noting food plants available to, or utilised by, the iguanas. The presence of feral cats, dogs or ungulates (burros, goats, cows), or rodents, was also determined using sightings, tracks, or presence of dung. We noted the occurrence of other species of native vertebrates on each cay as well. Through the use of these techniques, we selected Long Cay, Caicos Bank, an island that was large, previously supported iguanas, and had a population of feral cats. The island had no native mammals and no resident iguanas.

Feral cat eradication

To restore Long Cay we conducted a single intensive cat poisoning campaign on the island before any iguanas were relocated. We planned to augment the programme using leg-hold trapping if necessary. During this phase Dick Veitch, who had years of experience with cat control, joined the collaborative effort. 1080 (sodium monofluoroacetate) was chosen for the toxin because he had used it, along with leg-hold trapping, to successfully eradicate cats on islands in New Zealand (Veitch 1985). If cat problems recurred after iguanas were relocated to Long Cay, we planned to follow-up with live trapping as needed. Our group consulted with Charles Wigley, 1080 manufacturer (Tull Chemical Company, Oxford Alabama), who gave us additional guidance on dosage and handling.

Loose sand is a common substrate on Long Cay ridges, flatlands, and beaches. Cat tracks were evident in sandy regions when cats were present on the island. We used tracks as an index to cat abundance and to identify the areas they used. The poisoning campaign was conducted

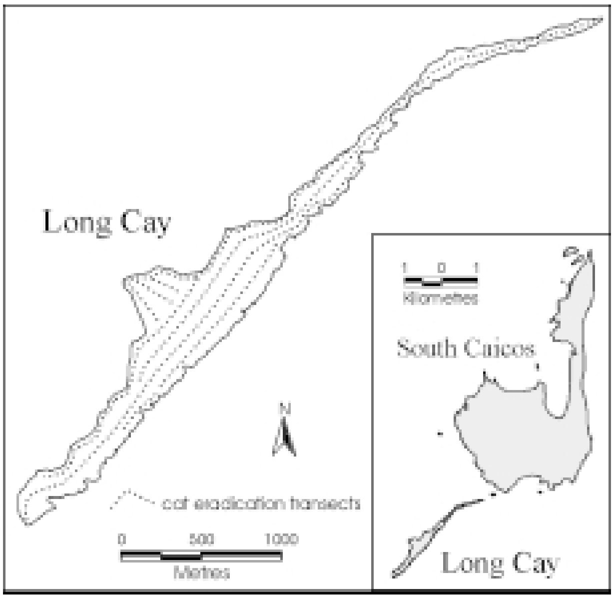

Fig. 2 Long Cay, Turks and Caicos Islands, showing transects along which 1080 bait stations were placed at 25 metre intervals to kill cats.

in July 1999. Although the cats seemed localised in certain parts of Long Cay, we set up bait stations that allowed us to systematically distribute the poison baits uniformly over the 3.5 km long island. Bait stations were marked with bright pink surveyor's tape or bright red plastic cups, numbered, and spaced 25 m apart in roughly-parallel lines 50-100 m apart. The north-east section of the island is less than 100m in width and therefore had only one line of bait stations; wider mid-sections of the island had four parallel lines of bait stations, and so on, depending on the width of the cay (Fig 2).

We used fresh whole minnows or fish chunks injected with 0.009 ml of a 22% 1080 solution for cat bait. The minnows (*Allanetta harringtonensis*, Atherinidae) or sprat (*Harangula* sp., Clupeidae) were seined daily in the morning. Larger fish were cut into 2 cm^3 sections. 1080 was injected into the peritoneal cavity of minnows and into the musculature of cut fish. Most of the bait was placed or skewered on branches overhanging clearings or trails at a height of about 15 cm. This suspended the bait at cat nose-height and out of the reach of land crabs. On the beach, or in areas without vegetation, bait was placed on inverted red plastic cups (15 cm high) that were filled with sand to prevent them from being displaced by wind. The cups also kept baits clean and away from crabs. Thorough and even coverage of Long Cay required 460 bait stations. We would, however, set up to 500 because we added stations along beaches, the areas most frequented by cats. Bait was laid at the stations between 1600-1900 hrs to minimise exposure to heat and scavenging birds. Old baits were collected when fresh bait was deposited daily for 5-6 days. At the end of the week, leftover toxin and contaminated items were diluted to non-toxic levels and disposed of or burned, respectively.

Iguana relocation and monitoring plan

Once cats appeared to be gone from Long Cay we began translocating iguanas from areas of Big Ambergris that were undergoing incipient development. We endeavoured to collect all that we encountered, with a body mass greater than 250 g. Iguanas were captured using two principal methods: (1) a 200 lb (90 kg). test monofilament noose tied at the end of a 1.5-2 m fishing pole, and (2) pulling animals by hand from rocky retreats.

After capture, iguanas were immobilised with loops of surgical tape around their fore and hind legs. An additional tape loop was placed over the mouths of all animals to prevent them from biting each other. Animals were placed in groups of five in cloth bags. These were placed in shaded locations while we captured our quota which varied between 20-100 animals/day. The bags were then loaded onto a padded section of flooring in the DECR boat in which they travelled to South Caicos. There, at the DECR Fisheries Laboratory iguanas were subcutaneously marked with PIT tags in the dorsal surface of the left thigh (allowing individual identification using a Trovan reader). The sex of all iguanas was confirmed by probe, animals were weighed, snout-vent length was measured, and distinguishing features were noted (e.g., regenerated tails, pigmentation, dorsal spine anomalies).

Iguana translocations from Big Ambergris to Long Cay usually occurred every 2-3 months. Generally, 10 individuals in each group relocated were fitted with 5 g radio collars (151 MHz) and were directly approached weekly and monitored until the next translocation.

During weekly checks of radio-tagged iguanas we recorded animal location using a differential global positioning system (DGPS), as well as habitat and behaviour variables. Most animals allowed us to approach them closely. Plant species within a 0.5 m radius of each animal were recorded. Most iguanas were seen basking on rocks, climbing in shrubs, or were hidden in burrows. At the end of 2-3 months, during the next relocation session, we would remove the collars and remeasure and reweigh the individuals before releasing them. We noted the condition of all radio-collared animals. As the numbers of animals on the island increased we caught and reported the condition and location of un-collared animals opportunistically. All information was transferred to a GIS database. The majority of the data collected will be detailed in a future report.

RESULTS

Relocation-site selection

We visited six cays that were previously known to support iguanas (Gerber 1995): Big Ambergris, West Six Hill, Middleton, Joe Grant's, Dickish, and one unnamed islet in the centre of Jackson Cut Bay (Fig. 1). Iguanas and their signs were usually spotted immediately. We saw no iguanas on Joe Grant's Cay when we visited parts of the eastern and western ends, but they were reported by Gerber (1995) to be rare there (approximately 50 individuals), perhaps patchily distributed, and we did not explore the entire cay. The numbers of iguanas on Middleton Cay seemed lower than the 150 Gerber (1995) reported in 1995; in two midday visits to the island a total of two iguanas and one tail drag were seen.

We re-evaluated nine other islands that were reported to have no iguanas present (Gerber 1995): Long Cay, East Six Hill Cay, J.A.G.S. McCartney Cay, East Caicos (Hog Cay, the south-eastern point, and Jacksonville areas), Sail Rock Island, one unnamed cay adjacent to Hog Cay, and three unnamed cays north of Joe Grant's Cay and west of Joe Grant's Point (Fig. 1). We attempted to visit Sand Bore and Big cays but could not reach them due to extremely low tides. We saw no iguanas on eight visited islands but found a previously-unreported dense population likely to consist of several thousand iguanas on the ninth, J.A.G.S. McCartney Cay (Mitchell *et al.* 2000).

In general, the small islands we examined that had suitable iguana habitat and no feral mammals already had existing iguana populations. We found that most islands without iguanas were either extremely small (<0.1 ha; iguana populations there would be small and vulnerable to extinction) or that they supported populations of feral cats or grazing ungulates (goats, cattle, donkeys, etc.) which suppress or eradicate iguanas. The latter would need restoration and management before they became suitable.

Therefore we concluded that we should focus our efforts on the largest cay with suitable habitat and on which feral mammals might successfully be controlled. Long Cay, Caicos Bank, was selected because of its size (approximately 3.5 km long, 111 ha), suitable habitat, and proximity to Big Ambergris Cay. Our visit confirmed the presence of cats previously reported by Iverson (1978) and Gerber (1995). Rats and mice (*Rattus rattus* and *Mus musculus*) also were in evidence on the cay but we did not consider them a threat to the iguanas as they occur on most islands in the Turks and Caicos, including those with healthy *C. carinata* populations. There were no other feral mammals on Long Cay, though goats and pigs had ranged there in the recent past.

Another important factor was that Long Cay was part of the former range of *C. carinata* (Schwartz and Thomas 1975). Iguanas had been seen there by John Iverson in visits between 1974 and 1977 (pers. com.), but had been since extirpated, presumably by resident cats.

We decided that the best alternative was to eradicate the feral cats on Long Cay, and focus the initial relocation effort from Ambergris Cay there.

Cat eradication

We found most of the evidence of cat foot traffic on Long Cay's beaches. Though large areas of the island contain sandy regions where tracks are easily seen, cat tracks were only noted twice away from the beach. The number of tracks we saw could be attributed to the wanderings of several individuals or small family groups. We did not attempt a population estimate but suspected there were fewer than 10 cats on the cay. The low number of cats present would facilitate cat removal. We conducted our baiting programme each evening from 8 through 12 July 1999. During the first three nights cat tracks were seen approaching three of our bait stations; two up on the limestone ridge, one on the beach. In those areas in which we did see cat tracks regularly, we did not see them during the last days of the study (11 and 12 July). No cat corpses were seen. We did not find any evidence of mortality in non-target species.

Weekly spot checks were made for cats on Long Cay during radio-tracking sessions in the following months. In early November 1999, we again surveyed the island thoroughly for tracks or signs of cats. In three days of walking surveys no evidence of cats was seen anywhere on the cay. No follow-up trapping appeared necessary. This result allowed us to proceed with the next step: iguana relocation.

Iguana relocation and monitoring plan

Until January 2000, during our field sessions and weekly radio tracking, no cat tracks were seen on Long Cay and survivorship of radio-tagged iguanas was 100%. During January and February 2000, radio-collared iguanas in the test group of 25 animals were recaptured and radio-collars were removed. Animals were weighed and measured. All recovered animals appeared healthy and each had established one or more burrow sites. Survivorship of this small group provided a second test for presence/absence of cats.

Later in January, we found tracks from a cat that one of us (Clerveaux) confirmed had been recently released on Long Cay by its owner from South Caicos. We succeeded in trapping and removing the cat from Long Cay within two weeks.

Since our first translocation in November 1999 we have relocated a total of 404 iguanas. We continue to collect data on habitat use, burrow location, home range dynamics, and will soon be collecting information on reproduction and recruitment of young. Since all Long Cay founders are PIT tagged, and dispersal to the cay is unlikely, we can be reasonably sure that untagged iguanas were born there. In January 2001, Mitchell confirmed the presence of two Long Cay hatchlings.

Condition

Repeat measurement of a random sample of 10 translocated animals (Fig. 3) shows that the average body weight increased by 252.5 g in the 3-12 months following translocation and that these animals were often larger than a random sample of 20 animals on Big Ambergris Cay at the same time of year (Fig. 3). The translocated sample includes four radio-collared individuals, two of which lost weight while the other two gained less than the average of the 10. If these radio-collared individuals are excluded from the sample, the average weight gain is 412.5 g. These weight observations on radio-collared animals suggest that either the collar or frequent human disturbance during radio tracking inhibits optimal growth.

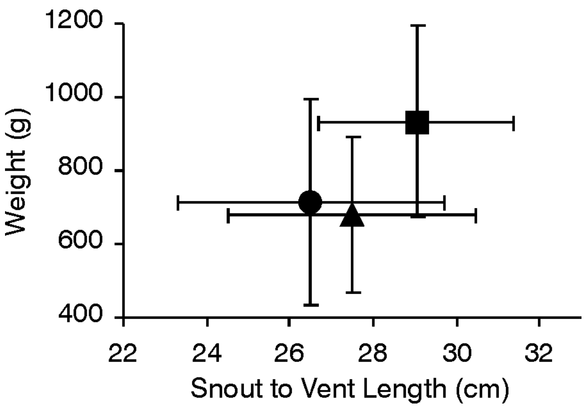

Fig. 3 Average weights and snout-vent length (cm) of a random sample of iguanas. Circle = 20 Big Ambergris Cay animals in January. Triangle = 10 Long Cay animals at the time of their release. Square = the same 10 Long Cay animals in January, 3-12 months after release. The error bars are ± one standard deviation.

DISCUSSION

Iguanas have disappeared from almost all of the larger Turks and Caicos Islands though some (e.g., Salt, Joe Grants, and Pine cays) still have relict populations (Gerber 1995). Most of these larger islands are overrun by feral hoofed livestock (donkeys, cattle, goats) or feral predators (cats or dogs). In his report to the National Trust, Gerber (1995) thoroughly described the negative impact feral mammals have on *C. carinata.* In particular, our survey reinforced the conclusion that, because cats are such efficient predators, cat populations cannot coexist with *C. carinata* (see also Iverson 1978).

For example, we surveyed three islands adjacent to East Caicos (50-100 m offshore) on which we found reasonably-dense populations of iguanas. We then conducted walking transects on the East Caicos shoreline opposite the iguana-populated cays. Though the habitat was vegetatively similar there was not a single iguana present. The most significant differences noted between East Caicos and the smaller satellite cays were the presence of cat and donkey populations on the larger cay.

Many of the smaller cays may be favourable for iguanas because they cannot support cats, the primary threat to iguanas. We suspect that every young iguana that disperses over water to colonise islands with an established population of cats will eventually be discovered and eaten. This is due primarily to the iguanas' small home range size, predictable behaviour, and sluggish movements in the colder early morning hours.

We suspect cats may not survive on small cays because of the general unavailability of fresh water. On large islands rainwater accumulates in the porous limestone rock. During periods of low rainfall, cats can reach this fresh water supply through solution holes and cavities in the rock. Smaller islands do not have fresh water reservoirs and thus do not provide long-term support for cats. In the absence of management, small cays and remote cays have been the salvation of the iguanas to date.

Populations on small islands, however, are more likely to fluctuate to zero (e.g., East Six-Hills Cay, Middleton Cay) and frequently they have impossibly-large dispersal distances between them. Satellite cays alone will not provide a long-range solution to maintaining the viability of *C. carinata*. Wherever it is logistically possible, it will be crucial to control or exclude exotic predators or competitors on large islands, and develop large, protected, island reserves. The larger, more stable, iguana populations supported will serve as genetic reservoirs and a source of dispersers (e.g., Long Cay, J.A.G.S. McCartney, Little Ambergris). We need to purposefully preserve and manage clusters of large and small satellite cays that together possess the qualities to ensure the survival of the endemic iguana.

The technique(s) required for managing invasive species on islands will probably be different in every case. Long Cay seemed well suited to the use of 1080. We carefully considered possible effects of the poison before conducting the cat removal work. Long Cay was unusual in that it had no iguanas, no native mammals, few scavenging birds, and no nesting colonies of scavengers. We judged that it was possible, but unlikely, that incidental bird deaths would occur.

Cats are extremely susceptible to minute quantities of 1080, 20 times more susceptible than humans, 10–30 times more susceptible than birds. Sub-lethal doses are metabolised and excreted. This chemical is broken down into non-toxic by-products by bacteria in soil or water. In freshwater the compound is 70% degraded after 24 hours (Veitch 1998). We also planned and equipped ourselves for supplemental cat trapping if it was necessary after the poisoning effort was concluded.

One of the things we found extremely encouraging at the outset was that, although much of the island has sandy exposed areas in which cat prints would be highly visible, in most areas no cat tracks were seen. Since we thoroughly explored all regions of the island before and during this baiting campaign, based on the few sets of tracks we saw, we feel that there were very few cats on Long Cay to start with (maybe 10 or fewer). It would have been difficult for these few cats to have avoided encountering a bait after we began the poisoning programme.

The equally-spaced, highly-visible, and numbered bait stations were useful for two reasons: (1) to assure good dispersion and thorough coverage of the island with bait, and (2) to allow us to find and recover uneaten baits. The first was important because it was vital that each cat on the island encountered a toxic bait during our programme. We therefore chose to have a small interval between bait stations: 8-12 baits per ha. The second reason that recognisable stations were important was that the toxicity of the baits rapidly degraded, as mentioned above. If uneaten baits were left available we risked saturating the island with non-toxic food sources for cats, which would decrease the likelihood of them eating fresh toxic baits. If baits were not placed at a bait station, uneaten baits would be very difficult to retrieve the following day because they were hard to see.

To keep Long Cay free of cats it is of the utmost importance to increasingly involve the community of South Caicos, reaching everyone with the message that unwanted animals should not be dropped off on uninhabited cays. DECR patrols and courtesy visits to vessels cruising the area are underway, and informative signs for Long Cay have been erected.

We have also begun production of a series of informative public announcements for local cable TV. In this regard we were lucky to have the services of a professional film maker, Vladimir Bibic. He donated time, equipment, and materials and completed two public service announcements that were aired in May 2000.

We will continue to monitor the establishment of *C. carinata* on Long Cay. Preliminary indications are that the Big Ambergris population is food limited, and that at this point Long Cay animals are not. The condition of recolonisers suggests that Long Cay iguanas are thriving and that the population has not reached carrying capacity. While the development on Big Ambergris advances we can therefore continue to relocate displaced animals until we have evidence that Long Cay is nearing population saturation. Based on observed *C. carinata* population densities elsewhere in the Turks and Caicos archipelago, and Long Cay's size, we expect that the island has the potential to support thousands of animals. We hope to use our experience on Long Cay to restore iguanas to other large islands in the Turks and Caicos where populations are currently dwindling or extirpated.

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Comparison of baits and bait stations for the selective control of wild house mice on Thevenard Island, Western Australia

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Abstract Past studies to eradicate or control house mice (*Mus domesticus*) have rarely been designed to reduce the impact on non-target and native species of rodents. General poison-baiting on an island reserve off the Western Australian coast required management actions to control or eradicate house mice in the presence of a threatened native short-tailed mouse (*Leggadina lakedownensis*). Cafeteria-style trials were conducted to ascertain a preferred bait medium that could be used to deliver a poison for house mice. When presented with a choice, the results show that it was not possible to make the level of bait uptake differ between the two species of mouse by treating the parrot seed with agar or wax, with or without the addition of salt to the bait. Three bait stations were tested for their effectiveness at selectively capturing house mice, or for the selective delivery of bait, and two showed promising results. From a management perspective, the use of these bait stations to deliver a poison bait for the control of house mice offers the most practical strategy without undue impact on non-target, native mice.

Keywords Australia; bait; house mouse, *Mus domesticus*; island; short-tailed mouse, *Leggadina lakedownensis.*

INTRODUCTION

Rodenticides are still the mainstay for the control of house mice. In Australia, these include strychnine, sodium flouroacetate (compound 1080), warfarin, brodifacoum, and bromadiolone, and they can be administered as liquids, powders, fumigants, gels, or baits (Rowe and Chudley 1963; Caughley *et al.* 1996). Baits are usually presented as attractive and edible foods, and include commercial pellets, wax blocks, coated cereal, or water (Caughley *et al.* 1996). Past efforts to eradicate or control house mice have not focussed on their selective control because this was not of primary importance during, for example, crisis management periods such as mouse plagues (Caughley *et al.* 1994), or they have been designed to exclude larger and non-target species such as birds (Taylor and Thomas 1993). Selective control becomes a concern, however, when the non-target species is also a rodent.

Compound 1080 is often used to manage invasive alien species in Western Australia (Mead *et al.* 1985). Its potential for target specificity is enhanced by a natural tolerance by many native species to the natural occurrence of the chemical in plants of the genus *Gastrolobium*, with which they co-evolved (McIlroy 1982). Fauna which have evolved in areas where these plants are absent are much less tolerant to 1080.

In Western Australia, house mice were introduced to Thevenard Island in 1986. Periodic plagues caused problems with the electrical facilitites of an oil storage and processing plant located on the island, and with the hygiene of work personnel. There was also concern that house mice would outcompete a rare species of short-tailed mouse. The presence of this native rodent made it unwise to broadcast spread a poison to control house mice. A study to identify the bait uptake and susceptibility to 1080 poisoning by short-tailed mice found they had a high projected intake of, and low tolerance to, this compound (Calver *et al.* 1989). Therefore, broadscale and non-selective baiting with 1080 on Thevenard Island were not options for the control of house mice, and other strategies for selective control were sought.

In an effort to identify ecophysiological differences between house mice and short-tailed mice on Thevenard Island, Moro and Bradshaw (1999) found that house mice had a higher requirement for water and sodium than the native species of mouse for the maintenance of physiological homeostasis. Since free water was limited to dew, which formed occasionally on the island, the main source of water for mice was from the plant and invertebrate material they ate. High water consumption by house mice is associated with their physiological need to meet high minimum-water requirements and to compensate for high water losses from evaporation. High sodium influxes were also observed for wild house mice, and reflected a dietary source in the field that was rich in sodium, in addition to a salt appetite at and above 0.25 $\mu g/l$ sodium concentrations (Moro and Bradshaw 1999). A requirement for high water influxes, and a taste for salt, may therefore offer a suitable means to control house mice selectively by exploiting their physiological needs for salt and water.

No published data are available that identify an effective bait medium or bait station to permit the control or eradication of the house mouse in the presence of a non-target rodent species. The delivery of poisons to rodents using palatable baits is a common strategy for population control (Sterner *et al.* 1996), and a study by Creekmore (1998) examined the effectiveness of administering biological markers to wild rodents using baits. It may be possible to exploit the house mouse's higher requirements for water and salt in the form of a palatable bait, and thereby formulate a selective poison bait while simultaneously reducing non-target mortality. I therefore evaluated the palatability of three baits, and relative preference of three types of bait station to deliver these baits, for the selective and future control of wild house mice on Thevenard Island.

METHODS

Source of mice and laboratory maintenance

Six adult short-tailed mice (three male, three female) and 12 house mice of various ages (six male, six female) were collected in early summer (December 1997) from Thevenard Island (21°28' S, 115°00' E). The island is a nature reserve situated 20 km off the north-west coast of Australia (Fig. 1), and experiences hot and humid summers and mild winters. A detailed description of the climate, vegetation, and geography is presented elsewhere (WAPET 1987; Moro 1997). Mice were air-transported within three days of capture to a controlled temperature room (air temperature = 25 ± 1°C, relative humidity = 40 ± 5%, 12:12 hour photoperiod) at Agriculture Western Australia (Forrestfield, Western Australia). Short-tailed mice were kept individually in plastic mouse containers (40 x 25 x 10 cm high). House mice were kept individually in glass aquaria (25 x 45 x 25 cm high) as they were less likely to escape when replacing food. All enclosures were secured with wire lids and supplied with paper as bedding material. Both species were acclimated to these enclosures for three days prior to the preference trials. Before trials, mice were maintained on an *ad libitum* diet of mixed parrot seed, and had apple available as a water source. Six individuals (three male, three female) of each species were used in each trial. The day before the first trial, the quantity of food given to each mouse was halved to encourage hunger. The same individual was used for each bait consumption trial so that comparisons were valid between trials. Each trial lasted for three nights.

Trial 1: Consumption of bait, no added salt

Three non-toxic media were chosen for the palatability trials: parrot seed, parrot seed coated with bees wax, and parrot seed coated with agar. These media were selected because they would be easy to procure for a broad-scale control operation, the coating would provide a suitable medium for a poison, and each bait could be produced in quantity. Baits of a known mass were presented separately and simultaneously in plastic trays (5 x 5 x 1.5 cm high) in a cafeteria format (Krebs 1989) to individual mice. Total mass of food supplied to each mouse during these trials exceeded their maximum intake of 2-3 g per day (Moro 1997), so bait was available at all times. The total mass of food consumed overnight was calculated (±0.1 g). An additional five of each bait type were placed in the room to measure evaporation overnight. Total mass of food consumed could therefore be corrected for any mass losses due to evaporation.

Trial 2: Consumption of bait, added salt

After three days, the experiment was repeated with the same cafeteria-style design, the same individuals, and the same bait and bait coatings (none, agar, wax), except each bait was mixed with salt. Aqueous solutions containing 0.25 μg/*l* sodium chloride were prepared in distilled water following laboratory trials that identified that house mice and short-tailed mice increased their water intake at this saline concentration (Moro and Bradshaw 2000). Equal volumes of saline were added to each bait coating, or to the seed (no coating).

Trial 3: Use of bait stations

The effectiveness of three bait stations was tested to evaluate their visitation by each species of mouse. Bait station one (BS-1) was constructed from a 20 *l* plastic bucket (Rheem, Australia; 40 cm high, 28 cm diameter; Fig. 2a).

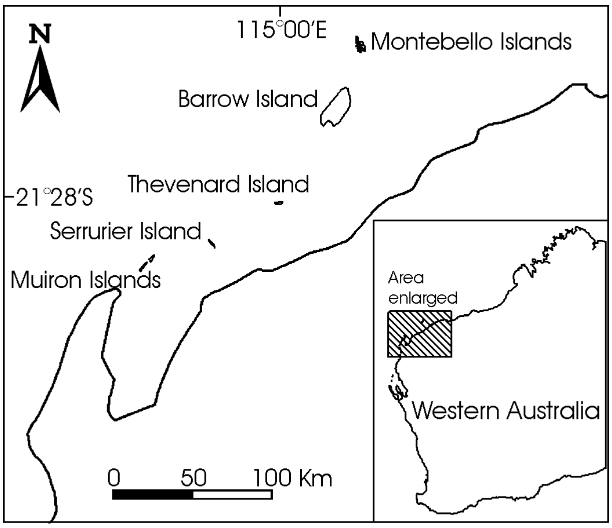

Fig. 1 Location of study site and surrounding islands.

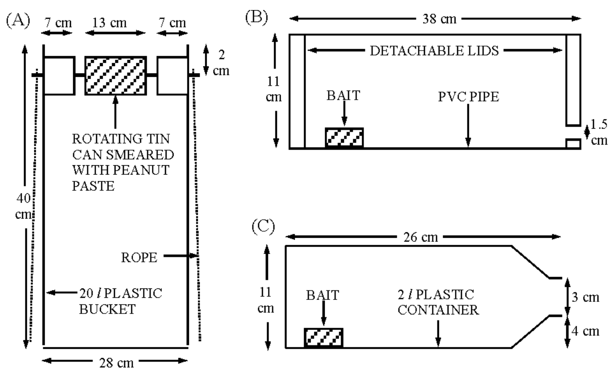

Fig. 2 Diagrams of the bait stations tested for the selective entry or capture of house mice: (A) BS-1, entry by ropes; (B) BS-2, entrance through minimum hole size; (C) BS-3, entrance raised and through a larger hole size.

It comprised a metal rod passed through three enclosed aluminium cans (7 cm high, 6.5 cm diameter), resting approximately 2 cm below the rim so that the cans lay neatly along the inside of the bucket. The cans on either side of the central can were secured to the bucket using an epoxy resin or silicon, while the central can (13 cm high, 6.5 cm diameter) was smeared with peanut paste and left to freely rotate. Rope (1.5 cm diameter) was hung on either side of the rod. This design is currently in restricted use on Thevenard Island. It works on the principle that a mouse will climb the rope, rest on the fixed can, and smell the peanut paste, whereupon it will move onto the central can which spins on the rod and causes the mouse to fall inside.

Bait station two (BS-2) was constructed from a PVC tube (38 cm long, 11 cm diameter) fitted with lids at each end, one of which was perforated with a hole that only permitted the entry of house mice (Fig. 2b). Bait was placed inside the tube at the opposite end to the point of entry. This design was dependent upon the use of a suitable hole diameter that excludes the entry of adult short-tailed mice. To identify a suitable hole size, adult mice of both species were individually placed inside a large aquarium (76 x 30 x 36 cm high) fitted with four perspex walls, and left overnight. Each wall was perforated with a hole of diameter 20 mm, 15 mm, 13 mm, or 10 mm. A small tray of food that was placed between partitions was disturbed if a mouse entered that partition. When the minimum hole diameter that would permit the passage of house mice but not short-tailed mice was found, it was drilled into one of the lids of the plastic tube and used in the trials. Each tube was positioned horizontally on the floor of the room with the hole close (0.5 cm) to the floor.

The third bait station (BS-3) was a 2 *l* plastic milk container (26 cm long, 11 cm diameter; Fig. 2c). The parrot-seed bait was placed inside the tube opposite the entrance, and the bait station was positioned horizontally on the floor. BS-3 differed from BS-2 because it was of simple design, and had a larger entrance diameter (3 cm) which rested higher (4 cm) off the floor.

All trials were performed in a controlled-temperature room, as described before, over a three-night period. Each bait station design was tested separately. Bait stations were positioned throughout the room, and mice were released and left overnight. The disturbance to the parrot-seed bait inside BS-2 and BS-3 was used to gauge mouse visitation. Mice captured within BS-1 could be counted before release. A total of 12 house mice were initially used during these trials. The experiment was then repeated using six short-tailed mice after all 12 house mice were removed.

Data Analysis

Manly's selection index, β (Manly 1995), was used to assess the difference in the amount of food eaten by each species when offered a choice. The β indices are estimates of the actual consumption as a proportion of the initial amount of food provided. The global 95% corrected *t* distribution confidence intervals (CI) follow equations in Manly (1995), and provide the limits to which the null hypothesis of no selection for a food type [where no selection(β) = 1/(number food types used)] can be compared. In all statistical analyses, a probability of $P \leq 0.05$ was considered significant. Total food consumption between trials was compared using a paired *t*-test after data were logarithmically transformed.

RESULTS

Trial 1: Bait consumption, no added salt

When presented with a choice of bait media, house mice selected parrot seed coated with agar 84% more often (global 95% CI 71-98%) than either parrot seed alone, or parrot seed coated with wax (Table 1, Fig. 3). This selection

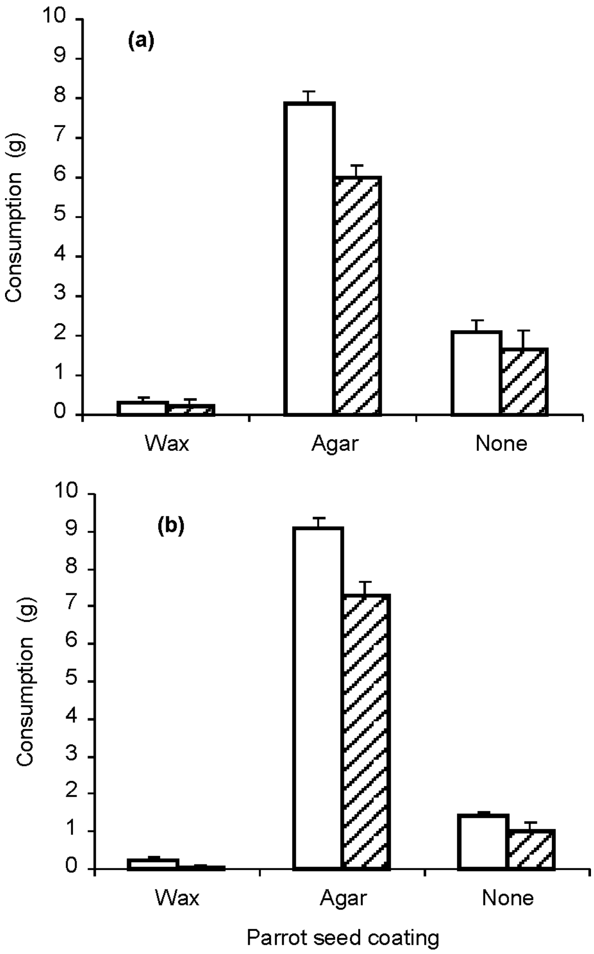

Fig. 3 Mean (±SE) daily consumption by short-tailed mice (open bars) and house mice (hatched bars) of bait (a) without added salt and (b) with 0.25 µg/*l* added salt. In each trial, consumption was calculated for six individuals of each species over a three day period, and only after each individual was acclimatised to their enclosure for three days. All three baits were presented simultaneously to each individual mouse.

Table 1 Results of cafeteria feeding trials for short-tailed mice and house mice offered parrot seed coated in wax (W), parrot seed coated in agar (A) and parrot seed alone (S). Statistics follow selectivity measures in Manly (1995), *n*=6 for each mouse species.

Trial		Short-tailed mice W	Short-tailed mice A	Short-tailed mice S	House mice W	House mice A	House mice S
No salt	β	0.02	0.85	0.13	0.02	0.84	0.15
	(±SE)[a]	(0.01)	(0.03)	(0.04)	(0.01)	(0.05)	(0.04)
	t(global)[b]	0-0.33	0.75-0.96	0.02-0.34	0-0.35	0.71-0.98	0.03-0.35
	Selection[c]	+/-	+	+/-	+/-	+	+/-
Salt	β	0.04	0.78	0.19	0.08	0.86	0.14
	(±SE)	(0.02)	(0.06)	(0.05)	(0.01)	(0.04)	(0.04)
	t(global)	0-0.11	0.54-0.92	0.08-0.31	0-0.03	0.71-0.96	0.04-0.27
	Selection	-	+	-	-	+	-

[a]Selection index (standard error)
[b]Global 95% corrected *t* distribution 95% confidence intervals
[c]Food selected for (+), avoided (-), or neither selected nor avoided (+/-)

is significantly different from the null hypothesis expectation of 33% if no food selection was detectable. Of the remaining foods, house mice selected parrot seed alone 15% (global 95% CI 3-35%) of the time, and parrot seed coated in wax only 2% (global 95% CI 0-35%) of the time.

The selection of bait type by short-tailed mice was similar to that of the house mice (Table 1, Fig. 3). Short-tailed mice selected parrot seed coated in agar 85% (global 95% CI 75-96%) more often than parrot seed coated in wax (2%, global 95% CI 0-33%), or parrot seed alone (13%, global 95% CI 2-34%). Clearly, parrot seed coated in agar was the preferred food type selected by house mice and short-tailed mice. However, parrot seed coated in wax, and parrot seed used alone, were neither avoided nor preferred by either species.

Trial 2: Bait consumption, added salt

When house mice and short-tailed mice were presented with a choice of foods after the addition of 0.25 μg/*l* salt to the foods, agar was still their preferred choice (Table 1), and consumption increased (Fig. 3). In contrast to Trial 1, both house mice and short-tailed mice showed an avoidance for parrot seed with or without a wax coating. However, when the type of bait is ignored, total consumption of food (mean ± standard error) increased after the addition of salt to the baits. The total food consumed by house mice was significantly higher (22.2±1.6 g/day, $t = 9.83$, df = 5, $P < 0.0001$) after salt was added to their foods than before added salt (7.9 ± 0.3 g/day). Similarly, short-tailed mice consumed significantly more food after the addition of salt (31.0±1.7 g/day) than before its addition (10.3 ± 0.5 g/day, $t = 12.1$, df = 5, $P < 0.0001$).

Trial 3: Efficiency of bait stations

The bucket design bait station (BS-1) captured five, seven, and 10 house mice in each of three consecutive nights, respectively. This design required that the captured mice had to be removed regularly.

The minimum hole diameter that all house mice were found to pass through was 15 mm (Table 2). Only one short-tailed mouse was found to pass through a hole of this dimension, although this individual was the smallest and lightest in body mass (19 g) of all short-tailed mice used in the trials. This hole dimension was subsequently drilled into the lid of BS-2 to identify entry by each species of mouse. Visitation to BS-2 and BS-3 was recorded for house mice.

In contrast, no short-tailed mice were captured using BS-1. One individual short-tailed mouse was found within BS-2 after it had enlarged the entrance hole by chewing. All BS-3 designs showed evidence that short-tailed mice had entered.

Table 2 Body mass of house mice and short-tailed mice collected from Thevenard Island, and frequency of mice that successfully passed through a hole of a specified diameter, as used during the bait station trials.

Species	*n*	Body mass (g) Mean (±SD)	Hole diameter (mm) 20	15	13	10
Short-tailed mouse	6	22.8(4.3)	6	1	1	0
House mouse	12	12.7(4.2)	12	12	5	0

DISCUSSION

The cafeteria trials showed that parrot seed mixed with an agar coating will increase bait consumption by both house mice and short-tailed mice. However, the level of bait uptake did not differ between species of mouse, indicating that, if used on its own, parrot seed mixed with an agar coating would not be a suitable medium to lace with a rodenticide for house mouse control without affecting the short-tailed mice.

The addition of salt to the agar increased the consumption of the bait by both rodent species. Adding salt to a bait could therefore offer one way to increase bait consumption by house mice. This increase could be an inherent need to consume moist foods (agar) to compensate for an increase of salt into their bodies. Alternatively, a more salty diet may have stimulated an increase in the consumption of agar because mice developed a salt appetite (Denton 1982), and is consistent with laboratory data demonstrating high saline intakes in both species (Moro and Bradshaw 2000). Native bush rats (*Rattus fuscipes*) have also been found to increase their water intake within six hours of increasing the sodium content of the water, indicative that a taste for salt developed in response to an increase in sodium concentration (Abraham *et al.* 1975).

The selection of food can depend upon how well it satisfies the nutritional requirements of a rodent (Murray and Dickman 1994, 1997), or upon its physical or chemical characteristics (Westoby 1977). Agar provides a moist medium relative to a wax medium or to parrot seed supplied without an additional coating, and may explain why it was selected for during these trials. When presented with a choice of seeds of variable water content, the sandy inland mouse (*Pseudomys hermannsburgensis*) and the kangaroo rat (*Dipodomys merriami*) selected those with a high moisture content (Frank 1988; Murray and Dickman 1997). Selection for moist foods is clearly an adaptive trait for a species inhabiting an environment where free water is scarce. The amount of water produced from metabolic processes can also be an important component of diet selection for mice (Olsen 1976; Post 1993). Alternatively, diet choice may be based upon caloric and nutritive factors as well. However, this is unlikely to have influenced the results of food selection by house mice or short-tailed mice in the present study, because each bait represented the same fundamental food type (parrot seed) presented with different coatings. Caloric (energy) content, therefore, would not have varied between bait media.

It must be recognised that the selection of a bait under laboratory conditions may differ to selection in the field, where the availability of alternative (and preferred) foods exists. For example, the effectiveness of strychnine in controlling house mice is well recognised (Caughley *et al.* 1996), but its effectiveness was found to be low when alternative foods were available in abundance (Brown *et al.* 1997). In the present study, parrot seed coated in wax was not a preferred bait by either species of mouse, perhaps because of an abundance of (preferred) seed coated with agar in the choice experiments. Elsewhere, wax has been used as an effective medium to deliver a poison. Rodenticide wax blocks (Talon™, ICI Australia) were effective for the control of house mice on Varanus Island, Western Australia (J. Angus, pers. comm.).

The results that investigate the suitability of a bait station for the control or capture of house mice selectively appear promising. Bait stations were successful at either capturing house mice (BS-1), or restricting the access of adult short-tailed mice to the baits (BS-2). Reasons why the short-tailed mice were not captured in BS-1 remain speculative, but may indicate a reluctance to climb. These designs exploit differences in the body sizes of each species (BS-2) and differences in their agility (BS-1), which may explain the reluctance of short-tailed mice to climb up the ropes and fall into the bait station. Buckets similar to BS-1 are currently in limited use around the dwellings on Thevenard Island, and to date have only captured house mice (West Australian Petroleum, unpub. data). The success of bait stations in restricting other species from accessing baits has been reported elsewhere. Bait stations made from plastic tubing have been used in New Zealand to poison rats whilst excluding birds (Taylor and Thomas 1993). The use of a bait box that partially encloses a rodenticide was found to be more effective for the control of house mice in a food store, compared to the use of the rodenticide alone (Rowe and Chudley 1963).

Management implications

Short-tailed mice face some risk of poisoning when poison is considered as a population management option to eradicate or control house mice, although the degree of exposure to toxic baits can be reduced. The use of an agar coating as a medium to deliver a poison for house mice seems a feasible option for use in the field. The addition of salt may increase the consumption of bait by a house mouse, and therefore increase the chance that an individual consuming a sublethal dose of poison will return to consume more. Agar coated baits are a feasible option for the delivery of rodenticides to wild house mice if non-target mice are absent. A preference of this bait and coating by short-tailed mice implies that these baits cannot be used for selective control if used on their own on Thevenard Island. However, exposure of a non-target mouse to a poison bait can be reduced if the bait is used in conjunction with a bait station such as BS-1 or BS-2. Plastic tubes that exclude short-tailed mice, such as BS-2, can provide a cost-effective method for broad-scale distribution to control house mice selectively. The use of tube stations on Thevenard Island will not restrict the entry of juvenile short-tailed mice. However, if their use is restricted to a time of year when juveniles are absent or low in density, they may reduce or eliminate non-target mortality and provide an effective control for house mice. Alternatively, a combination of the body size/agility selection process might make it possible to design a bait station that selected all house mice for all short-tailed mice. The combined use of a bait and bait station may therefore be an efficacious means to control house mice in areas where they coexist with non-target species of mice that may have threatened or endangered status.

ACKNOWLEDGMENTS

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The eradication of the black rat (*Rattus rattus*) on Barrow and adjacent islands off the north-west coast of Western Australia

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Abstract The black rat (*Rattus rattus*) has been introduced to many islands around the world and has been shown to have a detrimental impact on a wide range of fauna. It is known from about 1% of Australian Islands, of which many are adjacent to the Western Australian Pilbara or Kimberley coasts. Rats were accidentally introduced to these islands in the late 1800s by the pearling industry. Barrow and adjacent islands are nature reserves with significant conservation value, particularly for threatened mammals. Rats were known to inhabit the six smaller adjacent islands, but it was not until 1990 that they were located on the south end of Barrow Island. Eradication programmes on North and South Double, Boomerang, Pasco and Boodie Islands in 1983-1986 have been successful, but most of these islands had no non-target mammals. Seven mammals were considered to be at risk from an oat-based baiting programme on Barrow Island. Barrow Island was also considerably larger than other islands where successful eradication had occurred (23,000 ha vs 5 ha - 1000 ha). The rats on the smaller islands, without non-target mammals, were successfully eradicated using oats impregnated with the anticoagulant Pindone. Baits were laid on the ground in a 25 m grid. On Boodie Island unsuccessful attempts were made at covering the oat baits to prevent access by the threatened burrowing bettong (*Bettongia lesueur*). While the rats were eradicated, the bettongs also disappeared. They have since been successfully re-introduced and their abundance is well above pre-baiting levels. Fortunately on Barrow Island, the rats were present only in 245 ha at the south end of the large island. A bait station was designed that allowed climbing access by the black rats (and native rodents) but prevented access by other native mammals. These bait stations were set on a 25m grid throughout the area where the rats occurred. This eradication programme has been successful and the native rodents have since re-invaded the area. These bait stations were also used to eradicate rats on Middle Island where the threatened golden bandicoot (*Isoodon auratus barrowensis*) occurs. Abundances of golden bandicoots increased following rat eradication suggesting that rats may have suppressed bandiccot numbers. Monitoring of these reserves is continuing.

Keywords Barrow Island; black rat, *Rattus rattus;* eradication; non-target mammals; island management.

INTRODUCTION

The black rat (*Rattus rattus*) has been introduced to many islands around the world and has been shown to have a detrimental impact on a wide range of fauna, including native birds (Atkinson 1977, 1985; Taylor 1979; Moller 1983), reptiles and invertebrates (Ramsay 1978; Whitaker 1978). It is the most widely distributed introduced rodent on Australian islands, being recorded on 78 of the 8296 islands identified in Abbott and Burbidge (1995). In Western Australia it is known from 40 islands, most of which are near the Pilbara and Kimberley coasts. Black rats were accidentally introduced to many of these after the 1860s from shipwrecks, and the pearling industry, which made extensive use of the islands' bays for camping and careening vessels. Burbidge *et al.* (1997) did not find a relationship between the presence of rats and mammal extinctions; however, a more recent analysis (Burbidge and Manly 2002) does support such a link. Mammal declines and extinctions in the presence of rats include the burrowing bettong (*Bettongia lesueur*) on Boodie Island, nabarlek (*Petrogale concinna*) on Sunday Island, and the bush rat (*Rattus fuscipes*) and a native rodent (*Pseudomys* sp.) on Woody Island (Burbidge and Manly 2002). Black rats are also well-known as an exterminator of small ground-nesting seabirds - examples from Western Australia include the common noddy (*Anous stolidus*) and sooty tern (*Sterna*

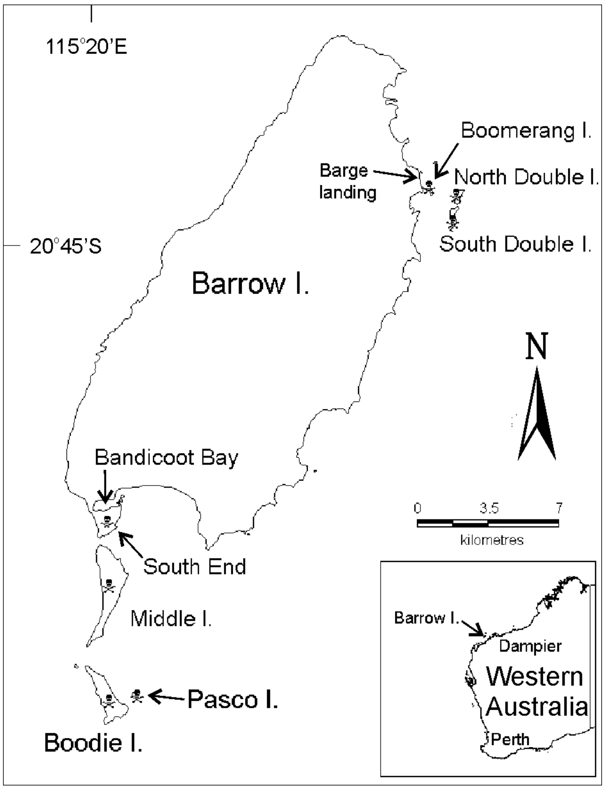

Fig. 1 The location of Barrow and adjacent islands off the north-west coast of Western Australia.

Pages 219-225 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

fuscata) from Rat Island (Houtman Abrolhos), and common noddy from Bedout Island and the Lacepede Islands (Tunney 1902; A.A. Burbidge pers. comm.).

Barrow Island and the adjacent six islands (Fig. 1) are nature reserves with significant conservation value, particularly for threatened mammals, and turtle and seabird nesting. Eleven terrestrial mammals are known from Barrow Island, with six of these listed as threatened species under the WA *Wildlife Conservation Act 1950* (Abbott and Burbidge 1995; Butler 1970, 1975). Three species of marine turtle and at least twelve species of seabird nest on Barrow and adjacent Islands. Seventeen species of migratory wader use the beaches and shallows for feeding and resting. In the 1890s the waters around Barrow Island were fished extensively for pearls and pearl shell, and the beaches and bays were used for camps and careening of vessels. Since 1964, Barrow Island has been a producing oilfield and up to 150 people now live and work on the island.

Black rats were first recorded on Double Island in 1918 (Whitlock 1918) and on Barrow Island in 1976 (Kitchener and Vicker 1981). However, the Barrow Island record was from a white-bellied sea eagle nest adjacent to Double Island and it was believed to have come from Double Island rather than Barrow Island. In 1983, an inspection of the islands adjacent to Barrow confirmed the presence of black rats on North and South Double, Boomerang (which connects to Barrow at low tide), Pasco, Boodie and Middle islands. Extensive trapping on Barrow along the east coast near Boomerang did not find evidence that black rats occurred on that part of Barrow Island. However, in 1990 a black rat was trapped at the south end of Barrow, near Middle Island. Given that there were pearling camps in this area (H. Butler pers. comm.), it is likely that the rats were introduced from careened pearling vessels 100 years earlier.

Following the successful eradication of black rats on Bedout Island in 1981, using pindone-impregnated oat baits (Morris 1989), an eradication programme commenced on the smaller islands around Barrow Island in 1983. The islands without non-target mammals (North and South Double Islands and Pasco Islands) were baited first. Boomerang Island, which did have brushtail possums (*Trichosurus vulpecula arnhemensis*) and golden bandicoots (*Isoodon auratus barrowensis*), was also baited at this time. Because of its low tide connection to Barrow Island, mammals can move between the two islands and readily recolonise the island after baiting. Boodie and Middle Islands, off the south end of Barrow Island had populations of threatened mammals (boodie (*Bettongia lesueur*) and golden bandicoot (*Isoodon auratus barrowensis*) respectively) that were at risk from any rat-poisoning programme. However, in 1985 an attempt at selective baiting of the rats on Boodie Island was undertaken. After rats were discovered on Barrow in 1990, immediate plans were made to develop a rat baiting strategy that excluded as many of the non-target mammals on Barrow as possible. Seven mammals were considered to be at risk from an oat-based baiting programme (Table 1). This paper describes the techniques used for rat baiting programmes on Barrow and adjacent islands, and the results obtained.

Table 1 The area of islands and names of non-target mammals on Barrow and adjacent islands (* denotes threatened species under the WA *Wildlife Conservation Act 1950*).

Island	Area (ha)	Non-target mammals
Barrow	23,590	*Isoodon auratus barrowensis**
		*Bettongia lesueur**
		*Lagorchestes c. conspicillatus**
		*Macropus robustus isabellinus**
		*Petrogale lateralis lateralis**
		Trichosurus vulpecula arnhemensis
		*Pseudomys nanus ferculinus**
		Zyzomys argurus
		Hydromys chrysogaster
Middle	350	*Isoodon auratus barrowensis**
Boodie	170	*Bettongia lesueur**
South Double	23	none
North Double	12	none
Boomerang[1]	5	*Isoodon auratus barrowensis**
		Trichosurus v. arnhemensis
Pasco	2	none

[1] Connected to Barrow at low tide.

METHODS

Barrow Island lies approximately 60 km off the Pilbara coast of Western Australia (Fig. 1) and has been a nature reserve since 1910. It is also covered by a Petroleum Lease issued to West Australian Petroleum (WAPET, now part of Chevron) that has been an operating oilfield since 1964. Boomerang, and North and South Double Islands lie close to the east coast of Barrow, near the barge landing (Fig. 1). Middle, Boodie and Pasco Islands lie close to the south end of Barrow Island. These islands have been nature reserves since 1975.

The bait used for rat eradication on all islands consisted of husked oats impregnated with the anticoagulant pindone (2 pivalyl 1,3-indandione) at the rate of 0.17 mg per oat seed (2.8 g per kg of oats). Each bait station consisted of 150 g of pindone-impregnated oats contained in a palm-sized plastic bag. However, the method of bait deployment varied depending on whether non-target mammals were present or not. These methods are described below.

North and South Double, Boomerang and Pasco Islands.

In April 1983, baits were laid on Boomerang Island on the ground in a 25 m grid pattern with no covering (approximately 16 bait stations per hectare). A small tear was made in each plastic bag to facilitate access by the rats. Baits were also thrown into the low coastal cliffs from a dinghy. Baits were inspected every day for a week and replaced if more than two thirds of the oats had been consumed. A similar method was used to bait North and South Double Islands in October 1983 and Pasco Island in May 1985. Monitoring of success was by track and scat observations on all these islands at various times since the baiting occurred. The most recent monitoring of North and South Double, and Boomerang Islands was in October 2000, and Pasco Island in November 1998. Trapping (10 Sheffield cage traps and 10 Elliott traps set at 10 - 15 m intervals for two nights) was also undertaken on Boomerang Island in October 2000. Monitoring of these islands will continue as part of the Department of Conservation and Land Management's (CALM) fauna monitoring programme on Barrow Island.

Because Boomerang Island connects to Barrow Island at low tide, there was concern that black rats may have colonised the adjacent coastal parts of Barrow Island. In April 1983, an extensive trapping programme was undertaken using small and medium sized Elliott traps. Two more-or-less continuous lines of traps were run along the coastal dunes and cliffs from Mattress Point (1 km south of the barge landing) to Ant Point (1 km north of the barge landing). A total of 960 trap-nights were set. Another 329 trap-nights were set around incinerators and the warehouses near the centre of the island where all equipment is stored after being landed on the island.

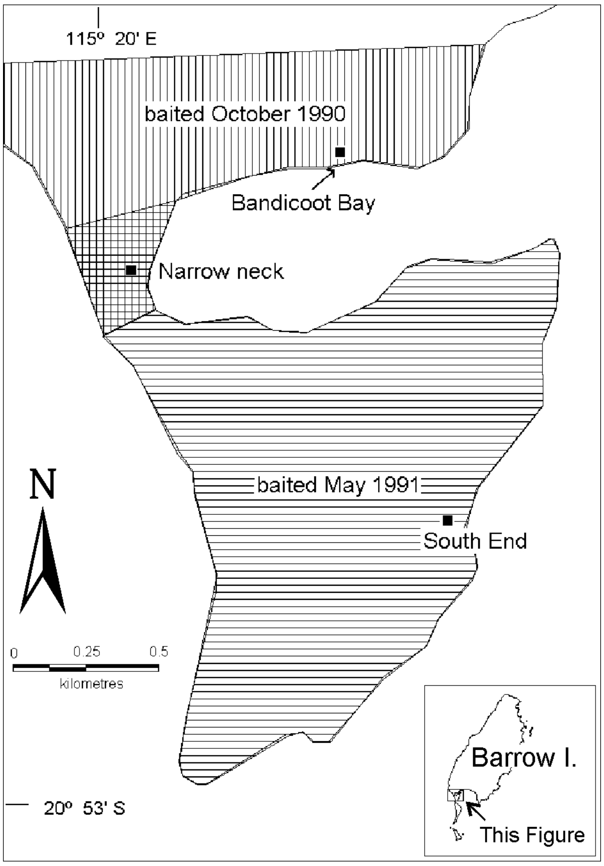

Fig. 2 The south end of Barrow Island showing baited area and the locations of trapping grids.

Boodie Island

The black rat baiting programme was undertaken on Boodie Island in May 1985. Baits were laid in a 25 m grid, however the presence of the burrowing bettong on a small part of this island necessitated covering some of the bait stations. We used an upturned plastic wash basin with access holes cut into the sides. These were designed to be large enough to allow access to rats, but small enough to prevent access by the bettongs. A previous survey of Boodie Island in October 1983 found that bettongs only occurred either on, or close to, the limestone outcropping at the south-east end of the island (approximately 15% of the 170 ha island). The 427 bait stations set in this area were covered. The remaining 994 bait stations were set uncovered in vegetated (*Spinifex longifolius*) areas on the remaining sandy part of the island. Large tidal inlets penetrate the north side of the island and baits were not set in these areas. All bait stations were checked daily for 10 days and baits replaced if necessary.

Three transects were established to monitor bettong and black rat numbers during and after the baiting. Transect 1 ran along the WAPET track from the beach and across the limestone to the Pasco # 3 oil well. Transect 2 ran along a sandy track between the limestone outcropping and a large sand dune to the north-west of the limestone. Transect 3 ran along the sandy beach on the north side of the island. Each transect was 500 m in length and it was walked by an observer with a head torch twice in an evening; once out and once back. The numbers of bettongs and rats seen on each leg of the transect were recorded and averaged. These transects were used again in September 1985, four months after the baits had been laid.

Trapping using 20 Sheffield cage traps and 25 Elliott traps set at 10-15 m intervals for two nights was undertaken in November 1998. In October 2000, 10 Sheffield cage traps set at 10 m intervals were set for one night.

Barrow and Middle Islands

Black rats were detected at the south end of Barrow Island in July 1990. In August /September the extent of the black rat distribution was determined and a bait station developed to reduce the take of poison bait by non-target mammals. In October 1990 a trial area of approximately 75 ha at the northern end of the known rat distribution was baited (to prevent possible colonisation of the rest of Barrow Island) with bait stations set at 25 m spacings. Using the same bait station spacings, the remainder of the south end (approximately 170 ha) was baited in May 1991, and Middle Island baited in September 1991 (Fig. 2).

The presence of large numbers of non-target mammals on Barrow Island necessitated the development of a bait station that allowed access by rats but prevented access by other mammals ranging in size from 30 g to 10 kg. The ability of rats to climb and to move through small holes was taken advantage of.

Table 2 Results of trapping along the coast of Barrow Island adjacent to Boomerang Island.

Species	Trapping Date / (# trap nights)					Overall trap success (%)
	19/4/83 (140)	20/4/83 (240)	21/4/83 (240)	22/4/83 (240)	23/4/83 (100)	
Isoodon auratus	29	45	43	34	12	17.0
Pseudomys nanus	4	17	17	23	5	6.9
Zyzomys argurus	0	1	6	8	3	1.9
Bettongia lesueur	8	4	4	4	3	2.4
Pseudantechinus sp.	2	0	1	0	0	0.3
Rattus rattus	0	0	0	0	0	0.0

The bait was enclosed in a 20 litre plastic bucket (Rheem) with 38 mm holes cut in the lid. Two external timber planks (90 cm long x 5 cm wide) ran from the lid to the ground. Two internal planks led from the holes in the lid to the bait on the sand (Fig. 3). The lid prevented access by the larger macropods (Barrow Island euro (*Macropus robustus isabellinus*), spectacled hare-wallaby (*Lagorchestes conspicillatus conspicillatus*), and burrowing bettong). The brushtail possum was prevented from reaching the bait by placing the poisoned oats on sand at least 13 cm below the lid. This was further than the maximum reach of a possum's forearm. The golden bandicoot was prevented from climbing to the top of the bait station by making the angle of the external planks greater than approximately 60 degrees. The native rodents (*Pseudomys nanus ferculinus* and *Zyzomys argurus*) could not be excluded from this bait station. However these species were widespread on Barrow Island and it was believed that they would be able to recolonise the baited area once the rats had been eradicated.

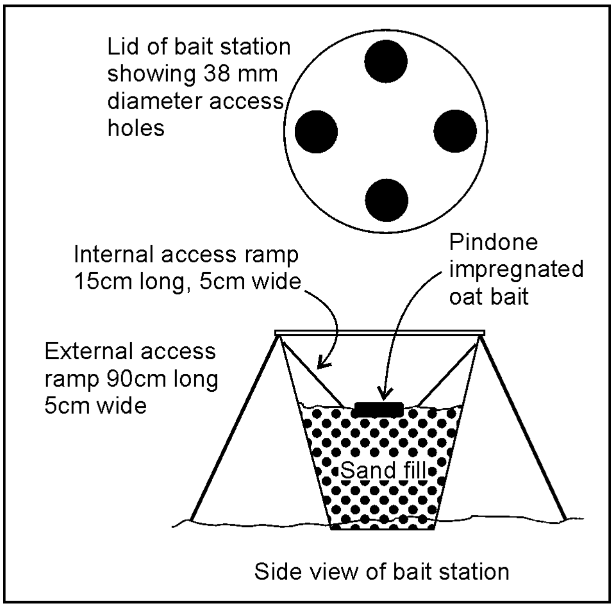

Fig. 3 Diagrammatic view of the bait station developed for black rat eradication on Barrow and Middle Islands.

Monitoring of baiting success was by trapping at three sites on the south end of Barrow Island (Bandicoot Bay, Narrow Neck and South End – Fig. 2), two sites on Middle Island, and searching for black rat tracks. On Barrow Island, each trapping site consisted of a 10 x 5 Elliott trap grid, with 20 m spacings between traps. In November 1998, the Bandicoot Bay site was changed to a 5 x 5 grid and cage traps and pit traps included at each trap point. Monitoring of this grid is ongoing as part of CALM's Barrow Island fauna monitoring programme.

RESULTS

North and South Double, Boomerang, and Pasco Islands

No rats were trapped on Barrow Island adjacent to Boomerang Island in April 1983 (Table 2) and none were detected around the warehouses.

Monitoring of Boomerang Island four months and six months after baiting (in August and October 1983) found no evidence of black rats, and this island has been visited opportunistically several times since. The latest was in October 2000 and no sign of black rats was detected. The brushtail possum and golden bandicoot have successfully recolonised Boomerang Island. Two possums were trapped in October 2000 and bandicoot tracks were seen on the island.

North and South Double Island were inspected 15 months after baiting (in February 1985) and again in September 1991 and no signs of rats were found. These islands were last visited in October 2000 and there was no sign of rats.

Pasco Island was visited four months after baiting (in September 1985) and again in October 1998 and no sign of rats was found.

Boodie Island

Within seven days of baits being deployed on Boodie Island, black rat numbers started to decline (Table 3), and

Table 3 The number of bettongs and black rats seen (average of two surveys) along 1 km transects on Boodie Island.

Date	Transect 1 (Limestone track) Bettongs	Transect 1 Rats	Transect 2 (Sand dune track) Bettongs	Transect 2 Rats	Transect 3 (Beach) Bettongs	Transect 3 Rats
20/10/83	2	26	-	-	-	-
18/5/85	0	20.5	1	2.5	0	0.5
19/5/85	4.5	26.5	0	1.5	0	1
20/5/85	1	23.5	0	2.5	0	1
21/5/85	1	12.5	0	1.5	0	1
22/5/85	3.5	14	0	2	0	1.5
23/5/85	2	14	0	1	0	1
24/5/85	2.5	4	0	0	0	0.5
25/5/85	3.5	4	0	0	0	0
26/5/85	1	3	0	0	0	0
27/5/85	2	1.5	0	0	0	0
28/5/85	1.5	1.5	0	0	0	0
29/5/85	1	1	0	1	0	0
17/9/85	1	0	0	0	0	0
18/9/85	1	0	0	0	0	0

subsequent visits have confirmed that the rats have been eradicated. Rats pulled some of the oats out from under the cover allowing access by the bettongs, which also declined and subsequently became locally extinct. In 1993, 36 burrowing bettongs were re-introduced from Barrow Island, and the population is now thriving. In the presence of black rats, the bettongs were restricted to the limestone portion of Boodie Island and there was evidence of rats sharing and competing for burrows with the bettongs. There were probably no more than 20 - 50 bettongs on the island when the rat eradication programme commenced. Trapping in October 1998 and October 2000 produced an 80% trap success rate for bettongs (no rats were trapped), and they have now colonised the entire island.

Table 5 Trap success rates (%) for golden bandicoots and black rats on Middle Island. Baiting was undertaken in September 1991.

	Sept. 1990	June. 1991	May 1992	Oct. 1992	Nov. 1998
Bandicoot	6.8	4.3	21.5	36.0	35.0
Black rat	3.0	1.8	0	0	0

Barrow and Middle Islands

The baiting programme undertaken in 1990/91 was successful in eradicating black rats from the south end of Barrow Island, and Middle Island. On Barrow Island, none have been trapped and no tracks have been seen since May 1991. Golden bandicoots and Barrow Island mice were also impacted and abundances declined during and immediately after the baiting programme (Table 4). However within 12 months, abundances of these native species had returned to pre-baiting levels. On Middle Island pre-baiting rat trap success rates were two to three times higher than on the south end of Barrow Island, suggesting higher rat densities on Middle Island (Table 5). Golden bandicoot abundance on Middle Island also increased five-fold following rat eradication.

DISCUSSION

An effective method has been developed for eradicating black rats in the presence of non-target mammal species on semi-arid islands up to 350 ha. The black rat eradication programme on Barrow and the adjacent islands has been successful and Barrow Island remains one of the largest landmasses on Earth without introduced mammals. This follows the successful eradication of black rats from the smaller Bedout, and Middle and West Lacepede Islands (Morris 1989; Abbott and Burbidge 1995). In other parts of Australia, only two other successful black rat eradication programmes have been reported (Abbott and Burbidge 1995) and these were on small islands of 1 ha and 42 ha.

Table 4 Trap success rates (%) for golden bandicoots, Barrow Island mice and black rats on Barrow Island. Rat control was undertaken at Bandicoot Bay and Narrow Neck in October 1990 and at the South End in May 1991 (- denotes no trapping undertaken).

	Bandicoot Bay			Narrow Neck			South End		
	Bandicoot	Mouse	Rat	Bandicoot	Mouse	Rat	Bandicoot	Mouse	Rat
August 1990	18.0	11.0	1.4	26.3	12.0	0.9	9.5	21.0	1.5
November 1990	8.2	0.5	0	8.7	0.7	0	-	-	-
May 1991	9.7	6.1	0	5.5	1.0	0	14.0	22.0	2.0
September 1991	21.6	4.2	0	29.2	5.8	0	-	-	-
October 1992	29.5	11.7	0	-	-	-	-	-	-
December 1993	28.0	14.0	0	33.0	21.0	0	27.0	31.0	0
November 1998	29.4	2.8	0	44.0	13.0	0	-	-	-
October 1999	36.5	5.5	0	-	-	-	-	-	-
October 2000	21.5	6.0	0	-	-	-	-	-	-

The work on Boodie Island was the first attempt in Australia to eradicate black rats in the presence of a threatened, non-target mammal. This study, and that of Short and Turner (1993), confirmed that the method used was successful in eradicating the black rat (its primary aim) but that the burrowing bettong was also eradicated. While this was not a desirable outcome in itself, the assertion by Short and Turner (1993) that this threatened the overall conservation of burrowing bettongs is not warranted. The estimate they provide of pre-baiting bettong numbers of approximately 20 was probably accurate, and it was very likely that the population would have become extinct if the rats had remained on Boodie Island. The baiting programme was undertaken by CALM because it was recognised that if all the bettongs were eradicated, it would be a relatively-simple operation to re-introduce this species from Barrow Island: this is what has occurred. The population of bettongs is no longer restricted to the limestone areas of Boodie Island and there are substantially more than the 20 individuals estimated pre-baiting, and substantially more than the 70 predicted by Short and Turner (1993) to be the estimated carrying capacity of the island. It is likely that as many as 200-300 boodies now inhabit the island. This substantial increase in abundance, and that of the golden bandicoot on Middle Island following rat eradication, supports the contention of Burbidge and Manly (2002) that black rats are implicated in declines and extinctions of mammals on islands.

It is interesting to speculate on the source of the black rats on Barrow Island and why the species was restricted to the sandy southern part of the island, about 1.5 % of its total area. Had it been more widespread, the eradication programme would have been far more difficult.

WAPET (now part of Chevron Australia) has been operating on Barrow Island since 1964 and, despite rigorous quarantine procedures, there have been occasional introductions of the house mouse (*Mus domesticus*) to the island. These have been quickly eradicated. Most of the oilfield activity occurs in the central part of the island, and most equipment is barged to the island, landing near Boomerang and North and South Double Islands. However, Whitlock (1918) reported black rats on Double Island in 1918, prior to the development of Barrow as an oilfield. Pearlers used the area now known as 'The Landing' in the 1890s as a campsite (W. H. Butler pers. comm.) and it is reasonable to assume that the rats on Boomerang and Double Island originated from careened pearling luggers. An extensive trapping programme along the coast adjacent to Boomerang and Double Islands in 1983 did not find any sign of black rats. It is likely that the rats at the south end of Barrow Island also originated from pearling luggers in the 1890s.

Why didn't black rats invade other parts of Barrow Island? A possible explanation is that ecological processes kept Barrow rat free. While the adjacent islands have few or no native mammals, Barrow has a more complex guild of fauna. Any rats landing on Barrow would have to contend with increased predation from perenties (*Varanus giganteus*), Stimson's python (*Morelia stimsoni*), mulga snake (*Pseudechis australis*) and golden bandicoots, and competition for food from two species of native rodent, a possum and three small wallabies. Bandicoot Bay at the south end of Barrow was also used as a pearling camp and rats on Boodie and Middle Islands and the south end of Barrow probably originated from this activity rather than the oilfield development. It is not clear why the black rats were only restricted to a small part of Barrow Island, despite having probably been on the island for 100 or so years. The south end is almost entirely sandy with sparse vegetation and generally lower numbers of native mammals and predatory reptiles. The reduced predation and competition may have allowed the rats to establish and maintain a low population size in this area, but they were not able to establish in the majority of the island which is composed of limestone and supports high densities of native mammals and reptiles.

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Eradication of introduced Australian marsupials (brushtail possum and brushtailed rock wallaby) from Rangitoto and Motutapu Islands, New Zealand

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Abstract In 1990 the New Zealand Department of Conservation began an operation to eradicate the common brushtail possum (*Trichosurus vulpecula*) and brushtailed rock wallaby (*Petrogale penicillata penicillata*) from Rangitoto and Motutapu Islands in the Hauraki Gulf of Auckland. The operation began with a 1080 aerial drop on Rangitoto Island, achieving an estimated 93 percent kill of possums and wallabies. This was followed from 1990 to 1997 by ground work on both islands to complete the eradication of both species. Methods used were trapping, cyanide poisoning, dogs and spotlight shooting. This was followed by several years of ground monitoring and mop-up operations. Aerial surveillance, using a Forward Looking Infrared (FLIR) camera, was also conducted on two occasions to detect surviving animals. A Differential Global Positioning System (DGPS) (a navigational aid) logged flight lines and animals sighted. This was then interfaced on video footage so that survey data could be displayed in real time. The hunting team and their dogs were expected to operate under the harsh conditions of Rangitoto Island's rugged volcanic terrain. There were successes and failures with the multiple field methods employed in this operation. Results from a recent survey have indicated that the eradication of an estimated 21,000 possums and 12,500 wallabies was achieved in the eight years of the operation. The eradication operation has been successful in restoring the previously degenerating *Metrosideros* forest on Rangitoto and Motutapu Islands.

Keywords Brushtailed rock wallaby, *Petrogale p. penicillata;* brushtail possum, *Trichosurus vulpecula*; animal pest eradication; DGPS, differential global positioning system; FLIR, forward-looking infra-red; 1080 poison; aerial poison operation.

INTRODUCTION

Rangitoto Island (2300ha) is a dominant feature of the landscape of Auckland, New Zealand, recognisable from many places in the city. In turn Rangitoto provides from its summit a panoramic view of the Hauraki Gulf and the Auckland metropolitan area (Fig. 1). The recent basaltic volcanic cone of the island, formed *c.* 650 years ago, supports unusual native plant communities with a high level of endemism (Miller *et al.* 1994).

The majority of the forest canopy on Rangitoto consists of a unique association of pohutukawa (*Metrosideros excelsa*), northern rata (*M. robusta*) and their hybrids (hereafter referred to as '*Metrosideros* forest'). Only two other volcanic islands in the world support *Metrosideros*-based communities at a similar successional stage (D. Bellamy pers. comm.). These are located in the Galapagos and Hawaiian Island Groups. The ecological significance of Rangitoto is reflected in its status as a separate Ecological District by the New Zealand Department of Conservation (Department of Conservation 1993).

Motutapu Island (1550ha) is immediately adjacent to Rangitoto (Fig. 1). It is an older landform and its landscape of rolling green pastures and coastal *Metrosideros* forest differs markedly from its neighbour.

Two herbivorous marsupial species from Australia were introduced to the two islands. The brushtailed rock wal-

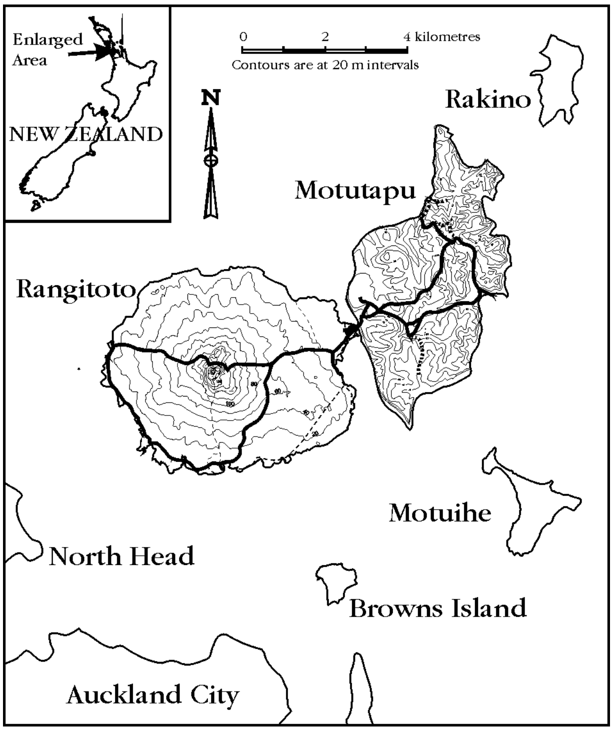

Fig. 1 Rangitoto and Motutapu islands and their proximity to Auckland City.

Pages 226-232 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

laby (*Petrogale penicillata penicillata*) was liberated on Motutapu Island in 1873 and from here the animals were able to move freely to Rangitoto at low tide. By 1912 they had reached high numbers on Rangitoto (Warburton *et al.* 1990). The common brushtail possum (*Trichosurus vulpecula*) was introduced to Rangitoto pre-1900 and to Motutapu Island in 1868 (Cowan 1990).

Browsing by possums and wallabies was recognised as early as the 1970s as a significant threat to Rangitoto Island's flora and fauna. The Hauraki Gulf Maritime Park Board (which administered the islands at the time) first made a request to the New Zealand Forest Service for advice on controlling possums and wallabies in July 1981.

The decline in forest health became more marked in the late 1980s with massive forest dieback, species loss and tree death common on both islands, all caused by severe mammal browsing (Department of Conservation 1990). About two-thirds of the *Metrosideros* forest had been so severely defoliated that it had died. As a consequence of the loss of vegetation cover, coastal cliffs were showing signs of severe erosion.

Previous control attempts had failed to contain the herbivorous mammal populations and decrease the browsing pressure. Since 1921 wallabies and possums had been controlled on Rangitoto Island using hunting and trapping. Population estimates in the late 1980s indicated possums had increased in abundance by 200% and wallabies by 33% since 1984 (Pekelharing 1991).

With this background the Hauraki Gulf Maritime Park Board resolved in 1987 to develop a work programme for eradication of possums and wallabies on Rangitoto and Motutapu Islands.

Since the islands are linked by a narrow causeway and bridge, any eradication operation had to cover the total area of 3850 hectares. Eradication of both possums and wallabies was established as the target because of the long-term benefits, relative to sustained control. Before the operation the Department of Conservation estimated there were 21,000 possums and 12,500 wallabies on Rangitoto and Motutapu islands (C. Pekelharing pers. comm.).

In 1990 funds were made available by the New Zealand Government for the possum and wallaby eradication programme to begin. Rotary International District 292 (Auckland) were major sponsors of the initial eradication phase which involved aerial poisoning.

METHODS

The Rangitoto-Motutapu Islands eradication operation started in September 1990, with employment of six full-time staff, including a supervisor. This Wild Animal Control (WAC) team was resident on the islands from Monday to Friday from 1991 to 1999.

Rangitoto Island

Several methods were used to eradicate marsupials from Rangitoto. These included aerial poisoning, cyanide bait stations, trapping and dogs.

Aerial poisoning

It was decided to apply cereal pellets containing 1080 (sodium monofluoroacetate) to kill possums and wallabies. Important factors in this decision were the rugged terrain and the need to quickly reduce the critical level of vegetation damage. Rangitoto Island is basaltic lava, much of which is jagged and loose, making it difficult and dangerous to traverse for ground hunting. Furthermore ground hunting can be difficult to control in terms of cost, time and performance (Department of Conservation 1990).

To meet public concerns over use of 1080 poison, the planned toxic loading in the bait was reduced to 0.08%, the lowest acceptable level for an effective target kill. Invertebrate and bird surveys were conducted which indicated a low level of risk to all species inhabiting the island; and the risk of contamination of the fresh groundwater lens was assessed as minimal.

In November 1990, after a series of public meetings and some very vocal opposition, 28.5 tonnes of 1080 pollard pellets (cinnamon-lured) were aerially distributed over Rangitoto Island. The sowing rate was 11.8kg per hectare and the true toxic loading was 0.073%.

This aerial operation was the first large poison operation to employ the assistance of a navigational guidance system for the helicopter. This was a Decca system working off triangulated radio beacons. The estimated kill-rate from this operation was 93%; approximately 20% higher than the average for aerial poisoning operations (Pekelharing 1991). This clearly demonstrated the benefit of navigational guidance for aerial baiting. This system also provided a method of auditing the aerial operation to determine the extent of bait cover on the ground (Fig. 2). These

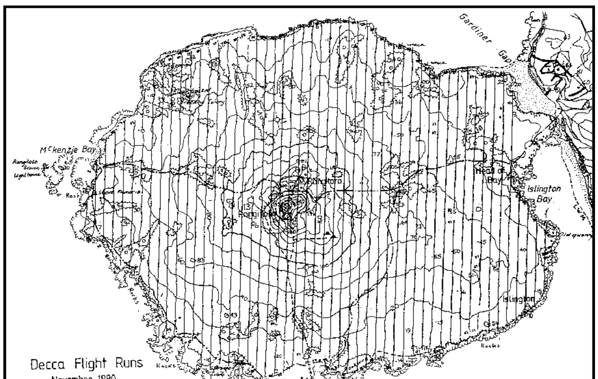

Fig. 2 Flight lines over Rangitoto Island during the 1080 bait drop in November 1999 as printed from the Decca navigation system.

systems have since been replaced by more effective Differential Global Positioning Systems (DGPS).

Cyanide bait stations

Paste containing cyanide poison (sodium cyanide) was laid in bait stations on Rangitoto Island from April l99l.

Animal activity at the bait stations gave a good indication of the location of residual animals. Even a few months after the aerial operation, animals were still exhibiting bait shyness. Bait shyness was evident in that some lines had to be pre-fed with non-toxic baits up to nine times to attract any possums or wallabies to the bait, before the toxin was applied. The pre-feed was lured with a fruit-based lure so animals would not associate it with the cinnamon lured 1080 poison.

Trapping

Initially, a trapping trial was conducted to identify the most effective trap available. At the time only Timms traps (KBL Industries, NZ) had been approved for humane reasons as a specific requirement of the major sponsor. Timms kill traps were compared with Victor No. 1.5 soft catch leghold traps (Woodstream Corporation, Lititz, P.A., U.S.A.). Victor traps were subsequently selected as they caught more animals and were more portable. Soft catch jaws were also used because of public pressure for humaneness.

From February 1991 Victor No. 1.5 soft catch traps were set out at 50 m intervals on 200 m interval parallel grid lines across the whole of Rangitoto Island. During wet winter days possums and wallabies often pulled out of the soft catch traps. Subsequent modifications failed to noticeably improve the performance of the trap set. After appropriate consultations a decision was made to change to steel jaw traps with Victor No. 1 leg-hold traps for possums and Victor No. 1.5 and No. 3 leg-hold traps for wallabies. The catch rate for both species increased markedly as a result of these changes (Department of Conservation 1992). However, it continued to be difficult to trap wallabies on Rangitoto Island given the lack of suitable sites for placement of traps.

For the 1992-1993 season the distance between grid lines was reduced to 100 metres for the second and third sweeps, with a total of 61 trap-lines being set across Rangitoto Island. A ‘rolling front’ method was used whereby traplines were leapfrogged into un-trapped areas, with 10 trap lines set at all times. Six lines were set within the new area and four lines remained in the trapped area as a buffer zone, to minimise any chance of animals moving back to the cleared areas. Hunters found that a large number of the possums trapped during these sweeps had previously escaped from soft catch traps.

Trapping on the 100 metre grid lines continued until mid-1996, when the tally for the year was down to one possum and 32 wallabies. Wallabies were targeted specifically along their preferred coastal habitat, with Victor No. 3 traps in double sets on the runs used by the wallabies.

Dogs

Dogs were used on Rangitoto Island in conjunction with the trapping operation. They worked along all trapping grid lines and where concentrations of animals had been found. Dogs proved invaluable for locating animals that had pulled out of traps and for locating fresh sign where additional traps could be placed. Possums that were found by the dogs were usually down holes in the lava and had to be dug out by hand. Wallabies were also located by the dogs, but only infrequently, as these animals were generally too fast for the dogs on the rough terrain.

Motutapu Island

The animal eradication programme on Motutapu Island focused on an immediate reduction in possum and wallaby numbers. A buffer zone approximately 1000 metres wide was cleared around the causeway between Motutapu and Rangitoto Islands to reduce the risk of r-einvasion. This was achieved using a combination of four methods – traps, poison, shotgun, and dogs.

Motutapu Island was divided into five blocks. Four blocks contained large coastal areas and the fifth block contained the conservation area in the centre of the Island.

Cyanide bait stations

Bait stations containing cyanide poison were used throughout all five blocks on Motutapu Island, from 1990 to 1992. Poison was laid in bait stations (a flower pot, or tin lid) with some conventional ground and tree baiting. As the cliffs were very steep, bait stations were set along the top of the cliffs at 50 metre intervals and, where possible, placed further down the cliff. All coastal areas and tree plantations were covered with the exception of an area left for trapping trial work. The poison was pre-fed a minimum of three times and lure was changed frequently to minimise bait shyness.

This method was successful in rapidly reducing the population of wallabies and possum. It was supplemented with extensive ground shooting. However, it proved difficult to target the two species concurrently. Wallabies fed during the day and removed bait before they were available to possums, which feed at night. This problem was overcome by pulsing (laying bait twice each day) the poison to target each species individually and by shooting wallabies before poisoning for possums.

Cyanide poison was effective in obtaining a quick knockdown of animals along the coastal areas and inland plantations. Combined with other methods, it relieved browsing pressure on trees along the coast.

Hunting

Initially the hunting of possums and wallabies on Motutapu Island was undertaken using three different methods: (a) dog and gun; (b) spotlight shooting; and (c) hunting in the evenings and driving the animals. To minimise interference with bait, all three methods were used to quickly reduce the wallaby population before a cyanide poison operation.

Spotlight shooting proved very effective for possum and wallaby control following the poison operations. It was also very effective in and around the small tree plantations on Motutapu Island, as the plantations were open and allowed for good animal recognition by shooters. Trapping would have been more effective, but at that time the operation did not have permission to use leg-hold traps.

In the coastal margins – and particularly along cliff edges – dogs and multiple hunters driving the animals was an effective method to quickly reduce the wallaby population. At times this resulted in more than 100 wallabies killed per hour. Shooting was undertaken using both 12 gauge shotguns and .22 calibre rifles. As the animal populations decreased, the use of dogs on Motutapu Island was important for checking all den and nesting sites.

The hunting operation had to be carefully monitored to ensure it did not interfere with other users of the island, including recreational fishers and boats.

Trapping

Victor soft catch traps (No. 1, 1.5, and 3) were used along the buffer zone once approval had been obtained, as the area had been poisoned four to five times and the residual animals were showing no interest in the bait. The trapping programme met with immediate success with initial lines removing the bulk of the remaining animals.

Initially, problems were experienced with wallabies pulling out of leg-hold traps. This was rectified by different approaches to setting the traps and the purchase of Victor No. 3 traps. Once an effective method of trap setting to target the wallabies had been found, the catch ratios doubled. The Victor No. 3 traps are a large trap and wallabies caught held well, with no pullouts. These traps were expensive but proved their worth as the animal numbers decreased and the risks of losing an animal through pullouts and trap shyness became more of a problem. Trapping in conjunction with dogs was the most effective method for total eradication.

The last possum and wallaby were caught on Motutapu Island in the 1993/1994 operational year.

Monitoring programme

Ground Operations

The monitoring phase of the eradication operation began in 1995. On Rangitoto Island, lines of leg-hold traps were set randomly at right angles to the original 200 metre grid lines. These monitoring lines were left in place and checked daily for eight weeks.

In addition, all grid lines were checked by the dog teams. Each grid line was checked for three days by each dog team, so that each grid line had nine person days of checks using dogs. This work was very hard on the dogs and turnover was fairly high. A dog would work well for a short period and then would lose interest in the work. However, if taken to the mainland for a short period to hunt where possums were common, the dog would work well again and with great enthusiasm.

Random lines were trapped three times and previous ‘hot spots’ were checked with traps and dogs. From 1994 to 1999, the hunting team monitored Motutapu Island using traps, dogs, and spotlighting.

Forward Looking Infra-red

Final monitoring of Rangitoto Island was completed by helicopter using a Forward Looking Infra-red (FLIR) camera to look for animals at night when the surrounding lava had cooled down. A Differential Global Positioning System (DGPS) navigational aid was interfaced to the FLIR so that video footage could be viewed and animal sightings plotted accurately using the DGPS information.

Table 1 Possum and wallaby kills on Rangitoto Island

Year	Possums per year	Wallabies per year	Trap Nights	Trap Nights per kill
1990	17,000[1]	8500	airdrop 93% kill [1]	
1990/91	182	10	180,000	937
1991/92	558	6	262,500	465
1992/93	268	17	239,800	841
1993/94	114	39	375,000	2450
1994/95	17	82	330,000	3333
1995/96	1	32	330,000	10,000
1996/97	0	4	240,000	60,000
1997/98	0	0	126,000	0
1998/99	0	0	42,000	0
TOTAL	1140	190	2,125,300	1598 *trap nights per kill*

[1]Estimated by Forest Research Institute

RESULTS

Possum and wallaby eradication

Tables 1 and 2 summarise the success of the operation on Rangitoto and Motutapu Islands from 1990-1999. The number of kills per trap night and the total kills for both target species indicate there was rapid population knock-down. This was followed by very long periods of time between kills. This had a significant effect on staff morale.

New Zealand Forest Research Institute (FRI) scientists estimated that the initial 1080 poison drop achieved a 93 percent kill, spread reasonably evenly over both target species (Pekelharing 1991). This represents about 17,000 possums and 8500 wallabies killed on Rangitoto Island during the first year of operation.

In the following six years annual possum kills ranged from 558 down to the final possum in 1996 (Table 1). Wallaby kills continued until 1997 when the last four were killed. On Rangitoto Island there was a total of 2.1 million trap nights, over nine years of eradication and monitoring. Catching the last possum and 32 wallabies in 1995/1996 required about 10,000 trap nights for each animal.

The main populations of wallabies and possums on Motutapu Island were found in the coastal band of approximately 400 hectares. The Motutapu Island tallies of animals killed include only the number of bodies picked up on some operations along the cliffs, but the hunting team estimated that up to 20 percent of the animal bodies were not recovered. Estimates from the ground operations during 1990 indicate that more than 3500 possums and 3500 wallabies were killed. Subsequent years saw the annual tallies drop off dramatically, with around 900 possums and 950 wallabies killed from 1991 to 1994 (Table 2). No further kills occurred and no animals were sighted from 1994 onwards.

Table 2 Possum and wallaby kills on Motutapu Island

Year	Possum[1]	Wallaby[1]	Method
1990/91	2989	3179	Poisoning, shooting
1991/92	660	637	Poisoning, shooting, trapping
1992/93	85	155	Shooting, trapping, dogs
1993/94	5	4	Shooting, trapping, dogs
1994/95	0	0	Trapping, dogs
1995/96	0	0	Trapping, dogs
1996/97	0	0	Trapping, dogs
1997/98	0	0	Trapping, dogs
1998/99	0	0	Trapping, dogs
TOTAL	3929	3768	

[1] It is estimated that up to 20% of the bodies were not recovered (Mowbray pers. obs.)

Two monitoring operations were flown over Rangitoto Island with the FLIR camera in 1997 and 1999. The first operation found two wallabies that were subsequently trapped using leg-hold traps. The 1999 survey was flown at a reduced height, enhancing the resolution of the camera. This second sweep found no sign of either wallabies or possums.

Since no animals were seen or caught in the extensive ground trapping monitoring operation (1995-1999) or during the final FLIR monitoring operation, both Rangitoto and Motutapu Islands are now declared free of possums and wallabies.

Effects of the eradication

The initial 1080 aerial operation also had an effect on the local rodent population which comprises Norway rat (*Rattus norvegicus*), ship rat (*R. rattus*) and house mouse (*Mus musculus*) (Department of Conservation 1990, Miller and Miller 1995). While rodents were not specifically targeted, they were reduced by the poisoning and there may have been a short-term benefit of reduced predation on bird eggs and chicks. The hunting team observed that if there had been a noticeable effect on the rodent populations that this period was very short, as by-catch of rodents was a problem with the trapping regime throughout the operation.

The eradication operation also resulted in a proliferation of several weed species (Wotherspoon 2002). This illustrates the need to anticipate and prepare for such changes following animal eradication projects.

Following the removal of possums and wallabies, rapid canopy and understorey recovery was evident on both Islands. Before the eradication of possums, heavy damage was inflicted on the young shoots and flowers of the *Metrosideros* forest on Rangitoto Island. Ironically, the increased flowering following possum and wallaby eradication was reflected in an increase in honey production from introduced honey bees (*Apis mellifera*). Bees may deplete the nectar resources available to the indigenous avifauna of the islands.

The Waitemata Honey Company has had beehives on Rangitoto Island since 1957 and has kept fairly accurate records of production. From the late 1970s to around 1985 production per hive ranged from 34 to 60kg. From 1986 honey production started to decline steadily and this was blamed on the damage to nectar-producing *Metrosideros* trees by possums. By the summers of 1988-1989 and 1989-1990, production was down to around 7-8kg per hive. After the aerial poison operation and the start of ground operations on Rangitoto Island, honey production rose to around

25kg per hive. This trend has continued, despite some variations that are attributable to other factors such as weather. The 1997-1998 production season saw a harvest of 81kg of honey per hive.

DISCUSSION

The people employed were the most important part of this operation. Although 20 different people worked during the entire operation (1990-1999), there was a core of staff involved in most of the eradication. Rangitoto Island is a hard environment to work in, with difficult terrain and extremes in temperatures due to the lava. Boots lasted only between four and six weeks on Rangitoto Island, as they were literally shredded from the hunters' feet by the lava.

As the eradication programme progressed, the low tallies affected motivation. So, periodically trips were taken to areas on the mainland where the hunters would have more successes.

Sustained motivation is one of the most important prerequisites for any eradication programme. A successful eradication operation must have consistent commitment from management and motivated staff to achieve the vision.

With hindsight it would probably have been easier to conduct the operation with a larger hunting team. This would have decreased the amount of time spent completing each sweep with the traps and dogs.

Use of FLIR for monitoring has the potential to become a very effective tool in eradication operations. However, there are a number of issues which need to be addressed when choosing to use FLIR. These include:

1. Availability of suitable systems (infra-red and DGPS) which are able to be integrated;
2. Operator experience in the use of DGPS (owner-operated or leased) and the ability to provide on-site print outs;
3. Ensuring the helicopter type is suitable for the operation and can operate effectively in the conditions. In the Rangitoto-Motutapu FLIR trial the camera was mounted to a Hughes 300. This was not an ideal helicopter for the operation as it had great difficulty maintaining the correct speed for survey while flying down wind, due to lack of power. During the actual FLIR survey the camera was mounted on a Hughes 500 and Squirrel helicopter, providing more stable all-weather platforms;
4. Helicopter operators must have sufficient time to set up the equipment for infra-red survey work.

The use of the FLIR camera by the New Zealand Department of Conservation is so far fairly limited, but work already undertaken has shown great potential for the monitoring or detection of a large variety of animal species. Infra-red monitoring has the potential to be a very cost-effective way of monitoring all animal populations.

The initial decision to use soft jaw traps, rather than steel jaw traps, added at least two years to the operation. The soft jaw traps were less efficient and the animals that escaped were often subsequently trap shy. Long periods of time were spent trying to catch trap-shy animals. One of the last wallabies caught had signs of being trapped at least three times previously. The decision to use soft jaw traps was made by management and not at an operational level. This decision could have compromised the whole operation and certainly added to the final cost.

Rock wallabies were a "wild card" for the whole trapping operation, as no work had been undertaken prior to the eradication operation, to ascertain if traps were an effective method to eradicate wallabies. The targeting of wallabies and possums at the same time also posed some problems. These were resolved, however, while the programme was underway. It may have been better if wallaby trapping trials had been completed prior to the operation commencing.

An opportunity exists to repeat the pre-eradication surveys of Rangitoto Island vegetation condition and avifauna (Miller and Anderson 1990; Miller 1992), to further quantify the apparent changes to the island ecosystems. The success of this operation as an exercise in ecological restoration is evident from the mainland in the proliferation of pohutukawa flowers and the visible greening of Rangitoto Island.

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An attempt to eradicate feral goats from Lord Howe Island

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Abstract Lord Howe Island is a 1455 ha World Heritage Site in the Pacific Ocean, about 700 km off the east coast of Australia. Feral goats became established soon after human settlement in 1834. Goats were removed from the northern part of the island in the early 1970s, but remained in the more rugged southern mountains despite efforts to eradicate them. A new plan to eradicate the goats was developed in early 1999 and an attempt made to do it later that year. This paper reports on how well the plan was matched by the operation. In the plan we used previous hunting tallies, kill rates, and guesses at rates of increase, to estimate that about 200 goats were present in 1999. To put all these animals at risk in one eradication campaign we estimated that both aerial hunting from helicopters (50 hours) and ground hunting with dogs (220 hunter-days) would be required, at a cost of NZ$107,000. The campaign began on 6 September 1999 and finished on 15 October 1999 during which time 295 goats were killed, 189 by the aerial hunting and 106 by the ground hunters. Eradication was claimed after the operation, but reports of fresh droppings and footprints were made in late 2000 and three goats were seen in 2001, one of which was shot in June 2001. Attempts by animal liberation groups to stop the programme, and a subsequent attempt to prosecute the hunters highlight the need for careful planning and management of animal welfare issues.

Keywords Feral goats; eradication; Lord Howe Island; animal welfare

INTRODUCTION

Lord Howe Island is a 1455-ha World Heritage Site located at 31°S 159°E in the Pacific Ocean about 700 km off the east coast of Australia (Fig. 1). The island was discovered in 1788 and has been inhabited since 1834. It has a resident population of approximately 350 people and can accommodate up to 400 tourists at any one time (Lord Howe Island Board 2000). Its native biota has suffered the usual catastrophes following invasion by alien vertebrates and weeds, with three species of birds being driven to extinction by the 1860s, and a further six species vanishing after the arrival of ship rats (*Rattus rattus*) in 1918 (Recher and Clark 1974). Feral pigs (*Sus scrofa*) and feral cats (*Felis catus*) were eradicated by the 1980s, and ship rats and house mice (*Mus* spp.) are currently controlled by poisoning in places, to protect the palm seed industry and seabird nesting sites (Eason 1996; Billing 2000; Billing and Harden 2000). Pickard (1984) lists 173 exotic angiosperms, with climbing asparagus (*Asparagus setaceous*), guava (*Psidium cattleianum*), and boneseed (*Chrysanthemoides monilifera*) being significant potential problems. Nevertheless, the rugged volcanic nature of most of the island (Mounts Gower and Lidgbird rise to 875 m and 777 m, respectively) has protected much of its natural biodiversity. Thus, the age of the island (7 million years), its recent occupation by humans (it was apparently never discovered by Polynesians), its protected status, and its remote location, has left Lord Howe Island as one of the least-modified of the few islands located at about this latitude in the Pacific Ocean.

Feral goats have been present since the island was settled. In the early 1970s, 228 goats were shot by the islanders in an eradication attempt. This was effective in the northern part of the island, although a single goat was still present on the northern cliffs in 1988 (D. Hiscox, pers. comm.). However, goats remained in the south; Pickard (1976) estimated that about 50 were present in 1975.

No assessment of the detailed effect of goats on the biota of Lord Howe Island was made before the eradication attempt, although browse damage to species (such as terrestrial orchids) was obvious in many places and regeneration of tree species appeared to be limited in areas where cyclones had killed adult trees (J. Parkes, pers. obs.). Even without detailed information, the ability of goats to alter insular ecosystems is well reported elsewhere (e.g., Coblentz 1978; Parkes 1984b) and a precautionary approach was justified.

The southern mountains are within the Lord Howe Island Permanent Park Preserve which has similar status to a national park. It is managed by the Board in accordance with a plan of management (NPWS 1986) prepared in consultation with the community. The plan of management requires a vigorous and regular shooting programme to be maintained, with the aim of eradication of the feral goats. The Lord Howe Island Board commissioned a report on the feasibility of eradicating the remaining goats (Parkes and Macdonald 1999), and this paper compares the predictions made in this report with the events that transpired in the actual eradication attempt.

Pages 233-239 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Fig. 1 Lord Howe Island showing the southern mountains inhabited by feral goats (photograph by Ian Hutton).

METHODS

Estimating the population size

Annual control of the goats in the south began in 1987, and between then and 1998, an average of 13.1 (range 7-28) hunting expeditions in which at least one goat was shot, were conducted each year. A total of 579 goats were shot with an annual range of 19-88 animals. The annual average kill-rate remained at between two and three goats killed per hunting expedition between 1987 and 1994, but thereafter increased to an average of 5.7 goats killed/hunting expedition (Fig. 2). If we make two bold assumptions, it is possible to estimate the population size from these data. First, we assumed that this increase in kill-rate reflected an increase in the goat population and not an increase in hunting efficiency and skills. The same individual was largely responsible for the hunting over the period so hunting ability was likely to have been constant, at least during the latter years. Second, we guessed at the rate at which the population would increase if all hunting stopped. We used a finite rate of increase of 1.45 based on some estimates from Australian mainland goat herds (Parkes *et al.* 1996). Simply, the number of goats present must have been able to at least double (the change in kill-rates) since about 1990 despite an average annual kill of 48.3 goats. We estimated the population to have been about 100 in 1987 and 200 in 1999.

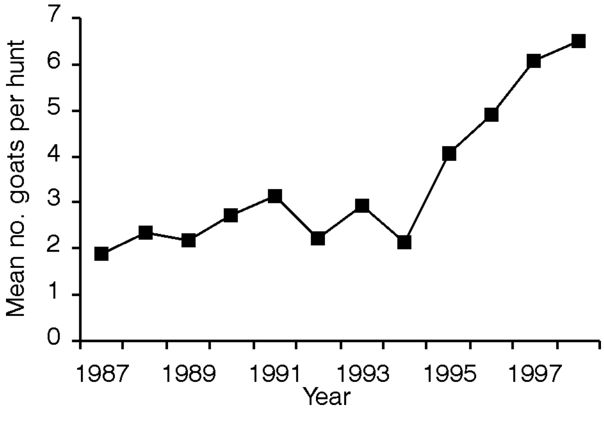

Fig. 2 Number of goats killed per hunting foray by Lord Howe Island rangers between 1987 and 1998.

Meeting the strategic rules for eradication

A policy of eradication but a practice of sustained control usually leads to sub-optimal results either because the failure to achieve eradication leads to the campaign being abandoned, or because the more difficult strategic requirements of sustained control (setting target densities and harvests to achieve protection while avoiding over-kill and opportunity costs) are not considered. So, the first considerations in any plan to eradicate must be its practicality and likelihood of success.

Parkes (1990) described three critical conditions that must be met before eradication is possible, and Bomford and O'Brien (1995) added three desirable rules that should be met for success:

- Rule 1: all the goats must be able to be put at risk.

This rule determines the tactics that must be employed. On Lord Howe Island, it was known roughly where goats were to be found, but it was likely that some goats would not be accessible to ground-based hunters and could only be killed using other methods. The use of dogs, aerial shooting from helicopters, Judas goats (e.g., Taylor and Katahira 1988), trapping, natural vegetation poisoning (e.g., Parkes 1983), and aerial poisoning (Forsyth and Parkes 1995) were all considered.

- Rule 2: the goats must be killed at rates faster than their rate of increase at all densities.

This rule determines the likely intensity and length of the campaign. On Lord Howe Island, there were thought to be only 200 goats, distributed over about 700 ha of forested habitat. Past experience suggested that a short, intensive campaign would be feasible, as well as desirable.

- Rule 3: the risk of re-colonisation must be zero.

Logically, because goats arrived once, this probability can never be zero. On Lord Howe Island, the risk is higher than zero as islanders keep a few domestic goats. Short of removing these animals, the plan recommended that: the risk that they could form a new feral herd could be reduced by banning anyone holding them on land adjacent to the forested reserves; setting fencing standards; keeping a register of domestic animals so that any escapees can be recaptured; sterilisation; or imposing a sunset clause on keeping goats (i.e., no new goats are kept as the current ones die).

- Rule 4: the social and economic conditions must be conducive to meeting the critical rules.

The Lord Howe Island Board was keen to eradicate the goats and had allocated the funds to do so. Not all the island residents were so convinced of the need. We estimated that about 36 hunter-days per year would have to be used in a sustained control campaign to hold goats at densities much lower than those in 1999. This would cost about NZ$6000 per year, as opposed to an estimated one-off cost of NZ$107,000 to eradicate the goats.

- Rule 5: where the benefits of management can be achieved without eradication, discounted future benefits should favour the one-off costs of eradication over the ongoing costs of sustained control.

This condition is only measurable when the benefits are accounted in the same currency as the costs; a difficult task for non-market conservation values and so not relevant for the Lord Howe Island case.

- Rule 6: ideally, goats surviving the campaign should be detectable and dealt with before an increased population size becomes obvious.

Failure to detect survivors increases the risk of operational failure. The plan recommended that two hunting sweeps in which no goats were seen should be a milestone to end the formal eradication campaign. However, the plan recommended some options to deal with any goats seen in *ad hoc* searches by local ranger staff after this milestone.

Tactical options

Selecting appropriate control methods requires consideration of the constraints imposed by the particular physical, biological, and social environment that might limit achievement of the strategic rules. On Lord Howe Island, the main physical constraints are habitat and topography. Most of the goat habitat is forested and often very steep, so neither an aerial campaign nor ground hunting would (by itself) be sufficient to kill all the goats over a short time. Other techniques such as poisoning were not suitable because of the lush vegetation, presence of non-target animals, or for social reasons.

The thick vegetation also meant that ground hunting without the use of dogs to find the goats would be, at best, inefficient. However, the use of dogs was itself constrained by the presence of the rare flightless rail, the Lord Howe woodhen (*Tricholimnas sylvestris*), so any dogs used would have to be trained not to molest non-target species. The programme also had to be scheduled so as to minimise potential impacts on native fauna, particularly seabirds which breed in the target area.

The main social constraint on the campaign was the presence of residents and tourists, the latter using many of the tracks to scenic areas in the areas inhabited by goats or taking guided tours to the summit of Mt Gower. This limited the campaign to a time of year when peoples' use of the area inhabited by goats was minimal.

The plan (Parkes and Macdonald 1999) recommended that a helicopter should be used to shoot all goats seen on the bluffs and ledges before the main ground campaign began. A Bell 206B (VH-HWS) owned by Helicopter Aerial Surveys Ltd was shipped from Australia to Lord Howe Island as deck cargo. The pilot was experienced in wild animal control operations, mainly on feral pig, goat, and buffalo control on mainland Australia. A Hughes 500C or 500D model would have been preferable in the windy conditions, but could not be arranged to meet operational deadlines. The helicopter was also used to ferry hunters and dogs during the ground hunting. To avoid any possible "incidents" with islanders and tourists, use of the helicopter over the whole island was constrained to the hours before 0800 h and after 1700 h, except for servicing the block being ground-hunted each day.

For the ground campaign, the plan recommended that a team of up to seven hunters using eight dogs in total would be ideal to cover a hunting block. The team hunting method used by the ground hunters has been developed by Prohunt NZ Ltd for use in New Zealand forested habitats where goats are difficult to find, do not usually associate in large groups, and where some usually escape any encounter with hunters or dogs (Parkes 1984a). The aim of the Prohunt system is to minimise the number of goats that escape death at the first encounter. The system uses a team of hunters each with one or two dogs that form a line across the area to be hunted with hunters being 100-150 m apart and in VHF radio contact with each other. The dogs are trained to chase and hold the goats, which are then killed by the nearest hunter. Dogs not immediately involved in the chase are trained not to join in so that the line is not broken, and the dogs doing the bailing are trained to return to their master once the goats are killed. The dogs used on Lord Howe Island had been trained to avoid kiwi (*Apteryx* spp.) in New Zealand. That training apparently worked for woodhen and nesting seabirds on Lord Howe Island, as the dogs ignored or avoided any encountered. No dogs spent any time "lost" during the campaign. Experience in New Zealand forested habitats suggested such a team could cover between 100 and 200 ha in a day, so the 700 ha area inhabited by goats on Lord Howe Island was divided into five hunting blocks. Again, experience from New Zealand campaigns suggested that most goats are killed in the first day of hunting, but up to six sweeps in each area would

Table 1 Goats killed, hunting effort and number of hunting sweeps at the end of the campaign when no goats were seen or killed in each of seven hunting blocks on Lord Howe Island.

Hunting block	Area (ha)	Number hunting sweeps	Number goats shot	Number sweeps no goats
Intermediate Hill	157	6	7	4
Boat Harbour	162	6	17	2
Erskine	168	10	31	4
Big Pocket	15	8	24	2
Mt Gower	30	3	0	3
Far Flats	190	4	5	2
World's End	188	11	22	2

Table 2 Predictions made in the plan and the actual data collected during the eradication campaign.

Parameter	Prediction from plan	Data collected during the operation
No. of goats	*c.* 200	295
Helicopter time (hours)	50	12.4 (hunting) + 19.7 (ferrying hunters)
Hunter days	220	168
No. hunters	7	6
No. hunting blocks	5	7
No. hunting sweeps/block	6	4-11
No. goats left	0	?

be needed to kill the last goat and give some measure of success (i.e., a sweep in which no goats were detected).

RESULTS

The Lord Howe Island Board endorsed the plan to eradicate goats in March 1999, and contracted Prohunt NZ Ltd to do the work. The campaign began on 6 September 1999 and ended on 15 October 1999. A total of 295 goats were killed, but goats were reported in 2000 so the campaign failed to eradicate the population.

During the aerial hunting 11 sorties were flown between 6 and 27 September (Fig. 3a) accounting for 189 goats in 12.4 flying hours. Most goats shot from the air (94%) were shot in the first five days. The ground hunters shot 106 goats, accounting for 61% of the goats shot from the ground in the first sweep, and 95% in the first four sweeps of the hunting blocks (Fig. 3b). Seven hunting blocks were used, their boundaries being natural features or tracks, and between one and four hunting sweeps when no goats were seen or shot were achieved in each (Table 1). The goats were surprisingly difficult to hunt for animals that had never been exposed to dogs, and could often outrun the dogs in areas of boulder fields. The ground hunters countered this by hunting towards the major bluffs so that escaping goats would take refuge there. The hunters then radioed the helicopter to deal with any seen.

DISCUSSION

How did the plan match the event?

We underestimated the number of goats present (Table 2). This was not surprising given the rough estimates we had for rates of increase and changes in kill-rates in the years before the eradication campaign on which the number was based. Only 64% of the helicopter time and 76% of the ground-hunting effort planned was expended because of budget limitations. As it turned out this was a mistake because eradication was not achieved in the initial operation. In late 2000, fresh droppings and footprints (of a small but unidentified number of goats) were found in one place under Mt Lidgbird. In March 2001, three goats (all apparently females) were reported at East Point at the northern end of the original range. The strategy to deal with these survivors was for the Board's ranger, who was involved with the original campaign, to hunt over the island

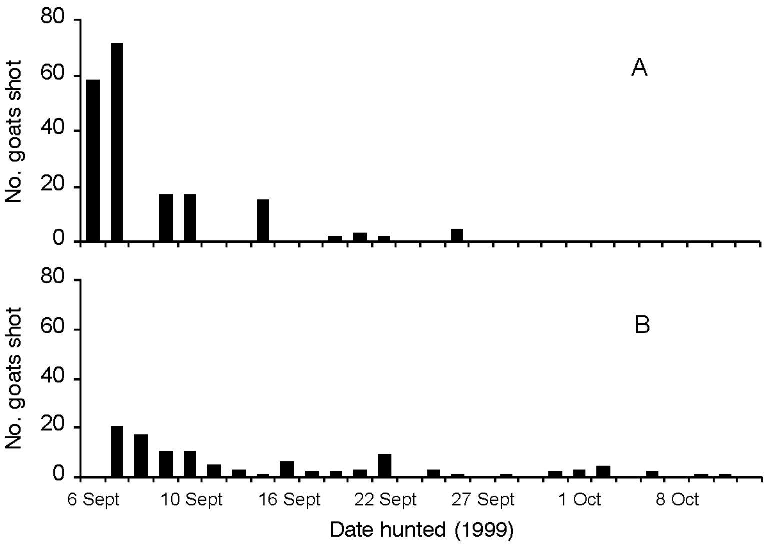

Fig. 3 Number of goats shot (A) in 11 helicopter sorties and (B) by ground hunters in the eradication campaign, 1999.

on one day each week. In June 2001, a goat that matched the description of one of those seen in March was shot about 4 km away on the western side of Mt Gower in an area with sign of a small number of goats (M. Carter pers. comm.). This area now appears to be the only place with goat sign (M. Carter pers. comm.) suggesting that only one small group of goats (perhaps now only two animals) is ranging over much of the former range of the herd. A Judas goat operation is planned for 2002 (S. Olson pers. comm.).

The main technical weaknesses in the campaign were largely caused by the fixed budget. The costs of transporting dogs from New Zealand meant that the hunters had only one spare dog to rest or replace injured dogs. The costs of the helicopter meant that the machine was not available for the last two weeks of the operation, with an increased risk that some goats might have survived on bluffs out of sight of the ground hunters, and the campaign lacked any systematic post-operational monitoring to detect survivors. In part, the ground-hunting control technique, with repeated sweeps of an area and careful recording of animals that escaped, gives some measure of success or failure, although the hunters thought another two weeks of hunting would have made them more confident of success.

More structured systems to detect low-density populations could be used if a high degree of certainty of success was essential. Such systems could be based on search theories developed as part of anti-submarine warfare (Koopman 1980) and now often developed to optimise search and rescue operations, or the presence of dangerous diseases in wildlife (Cannon and Roe 1982). The basic parameters in the theory are the need for coverage at an intensity related to the known or expected density of the items being sought, and to the detection range of the method used. Of course, in the case of presence/absence as in an eradication campaign or missing person search, the coverage must be complete, and some form of systematic sampling is required. It is feasible to search for (mobile) goats or their (stationary) sign over a small island such as Lord Howe, and detect their presence with high certainty, but very high intensity sampling is required to detect a survivor with a high degree of confidence somewhere in a large area by direct sampling methods (e.g., Choquenot *et al*. 2001). The Judas goat method is one way to increase the search range and confidence of detection for animals such as goats, and it would be worth exploring the optimal search parameters for a large eradication campaign such as Project Alcedo on Islas Isabela in the Galapagos Islands (Anon 1997).

Socio-political and animal welfare issues

The local island community was consulted during preparation of the plan to eradicate the goats and again prior to the Board's endorsement of the plan. Whilst there was generally strong community support, not all island residents were convinced of the need.

A group of island residents with previous experience in feral pig and goat control expressed interest in undertaking the eradication programme and, together with Prohunt NZ, were invited to bid for the work. However, the local consortium were unable to undertake the aerial shooting, which was considered an essential element of the programme. Nevertheless, provision was made for one local resident, who was also a member of the local consortium and on the Board's staff, to assist Prohunt NZ with the eradication operations.

Prior to and during the project, the Board liaised with the Australian Royal Society for the Prevention of Cruelty to Animals (RSPCA) to ensure that animal welfare considerations were properly addressed. Specific conditions were included in the contract to ensure that all goats were destroyed in a humane manner in accordance with legislative requirements and applicable codes of conduct. Board officers monitored operations in the field to ensure they were conducted to the required standards.

The Board distributed information about the eradication programme to local residents and visitors prior to and during the operation. This addressed the need for the programme, how it would be undertaken, the potential benefits, area closures, and other safety provisions. Regular progress reports were also provided to the community through the local island newspaper and one of the major Sydney daily newspapers ran a story on the project. This information assisted in raising community awareness, understanding and support for the eradication programme, and assisted to ensure that people did not enter the area during hunting operations.

Two issues arose. The first related to the proposed extent of the areas closed to the public for operational and safety reasons. Following representations from tourism operators, the proposed closures were revised and limited to the specific areas where eradication operations were being undertaken each day. This restricted operational flexibility and necessitated higher levels of planning and management, but did not adversely affect safety or the overall operation.

The second issue related to animal welfare concerns and disagreement about the need to kill the goats. Immediately prior to the programme commencing, a small number of island residents, including at least one associated with the unsuccessful bid for the work, collaborated with voluntary animal welfare groups to lobby for the programme to be stopped. Australian and international animal welfare organisations made numerous political and other representations and disseminated information over the internet. The major focus was on aerial shooting. However, there was also considerable public support for the programme from on the island, the Australian mainland, and overseas, much of it an apparent reaction to the emotive and misleading information distributed over the internet by the animal welfare groups.

Allegations of cruelty were referred to, and investigated by the local police and the RSPCA, neither of whom were able to establish any evidence to substantiate the allegations. Despite this, a private prosecution alleging offences against the (NSW) Prevention of Cruelty to Animals Act, 1997 was initiated by Animal Liberation New South Wales almost six months after the programme was completed. These were later withdrawn and costs were awarded against the prosecution.

Although animal welfare was carefully addressed in the planning and operational phases, and the operation inspected by animal welfare agencies, the level and manner in which this issue was pursued by animal liberation groups was not foreseen. In retrospect, this opposition was not surprising given past experience elsewhere in the world (e.g., during the goat and sheep eradication campaign on San Clemente Island off California (Van Vuren 1992)), and more recently against feral horse control in Australia (English 2001). No significant disruption to the programme was caused although some additional costs were incurred, mostly relating to the failed prosecution. We believe this signals an increasing interest by animal welfare organisations in invasive species control programmes which may have implications for other areas. It highlights the need for careful planning and management to ensure that control programmes are demonstrably needed to protect indigenous values, and are conducted in a humane manner consistent with legislative requirements and the highest standards of animal welfare. However, we note that despite the careful and skilful conduct of the control operation, adherence to standard operating procedures laid down by the Australian Government for aerial control of alien animals, the lack of evidence to the contrary, and the failure of the litigation, animal welfare groups are still using the Lord Howe case as an example of inhumane management and absence of evidence of the need for control of unwanted animals (Seymour and Oogjies 2001).

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Red mangrove eradication and pickleweed control in a Hawaiian wetland, waterbird responses, and lessons learned

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Abstract Alien red mangrove (*Rhizophora mangle*) and pickleweed (*Batis maritima*) are major invasive plants in Hawaiian wetlands, including Nu'upia Ponds, a 195 hectare wildlife management area and historic Hawaiian fishpond complex on U.S Marine Corps Base Hawaii. These fishponds are also home to approximately 10% of Hawai'i's endemic and endangered black-necked stilt (*Himantopus mexicanus knudseni*) population and at least 16 species of native fish. Invasive plants were changing the ecology and character of the fishponds from Hawaiian to Floridian. After 20 years of effort with thousands of volunteer hours, and over USD 2.5 million of contracted labour, over 20 acres of mangrove were removed. Mangroves were cleared by hand, shovels, and chain saws in archaeologically-sensitive areas and grappled with heavy tracked equipment in less-sensitive areas. Work was performed in the non-nesting season of the resident waterbirds. Prior to cutting, mature mangrove stands had been colonised by black-crowned night-herons and cattle egrets, causing work schedule alterations and the need for hazing permits. Pickleweed, an invasive ground cover, is annually plowed using Amphibious Assault Vehicles during "mud ops" training manoeuvres. The results show that stilts readily colonise mudflats cleared of alien vegetation, especially near established breeding areas. Lessons learned regarding waterbird conservation are discussed.

Keywords Hawaiian stilts; cattle egrets; black-crowned night-herons; egg measurements; red mangroves; pickleweed; tilapia; wetlands.

INTRODUCTION

Introduction of alien species has substantially changed the lowland landscape of Hawai'i. Alien plants displace native Hawaiian coastal plants, colonise unexploited habitats, trap sediments, and adversely affect water quality and hydrology. Alien animals consume primary producers, eliminate vegetative cover, foster erosion, and prey upon endangered species. Hawai'i's coastal wetland areas have been extensively altered for aquaculture, agriculture, grazing and urban development (Cuddihy and Stone 1990). Consequently the remaining Hawaiian wetlands that still harbour a few adaptive indigenous species also face a constant onslaught of alien species encroachment from the surrounding, extensively-altered landscape.

Hawai'i's intertidal wetlands in pre-contact times had only a few species of plants. Polynesian settlers, who altered many of these areas to plant introduced taro (*Colocasia*), grow seaweed, create saltpans, etc., affected plant succession. Egler (1939) suggested the following successional stages have occurred in many Hawaiian intertidal areas after Western discovery: (1) Original native communities of widgeon grass (*Ruppia maritima*), various algae and sessile organisms, (2) introduction of pickleweed (*Batis maritima*) and subsequent development of pure meadows, (3) introduction and spread of red mangrove (*Rhizophora mangle*), (4) extirpation of indigenous hau (*Hibiscus tiliaceus*) forests by mangrove forests, and (5) the eventual displacement of pickleweed meadows by mangrove forests (Allen 1998; Simberloff 1990).

Interaction of alien species in Hawaiian coastal wetlands has received little attention in the past (Cuddihy and Stone 1990). Few areas have been as well studied as Nu'upia Ponds. This paper reports on the 52-year relationship between two alien plants and four species of waterbirds on Mokapu Peninsula, O'ahu, before, during and after extensive alien plant control and eradication.

STUDY AREA AND SUBJECTS

Nu'upia Ponds Wildlife Management Area (WMA), under jurisdiction of the U.S Marine Corps since 1952, is located on the 1194 ha Mokapu Peninsula, one of several land parcels comprising Marine Corps Base Hawaii (MCBH). This peninsula separates Kane'ohe Bay from Kailua Bay, and the Nu'upia Ponds connect the peninsula to the rest of the island of O'ahu, Hawai'i (Fig. 1). The 195 ha complex today includes eight interconnected shallow ponds, associated mudflats and scrublands (Drigot 1999). Prior to Polynesian settlement, the ponds were thought to be either a shallow open channel between Kane'ohe and Kailua Bays, making Mokapu an island, or an embayment off Kane'ohe Bay with Mokapu connected to O'ahu by a thin coastal barrier dune-land strand. In either case, the Hawaiian settlers exploited this shallow open water area by subdividing it into several fishponds and a saltworks area, separated by hand-built coral and basalt rock walls. Later, 20th century settlers further subdivided these ponds by additional causeways into the eight ponds present today. Late 19th and early 20th century cattle grazing over most of the Mokapu Peninsula contributed to erosional sedimentation and creation of extensive mudflats

Pages 240-248 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

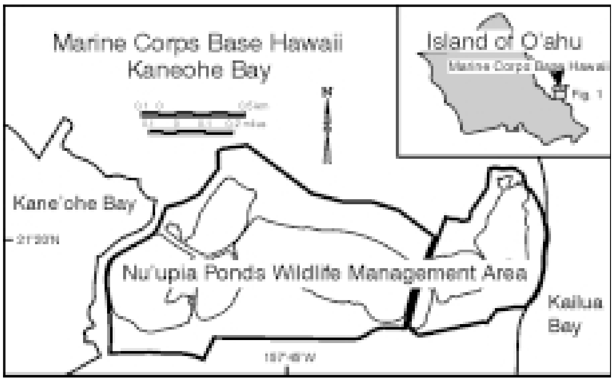

Fig 1 Location of Nu'upia Ponds WMA on Mokapu Peninsula, O'ahu, Hawai'i.

that have been largely colonised by alien vegetation. About one metre of very fine particle mud sits upon a solid and contiguous underlying ancient coral reef formation.

The shores of Nu'upia Ponds are bordered by thick, low vegetation mats composed almost entirely of pickleweed, introduced from South America to Hawai'i around 1859. The plant is highly salt-tolerant and grows in moist soil and shallow water. Short but dense monotypic stands of pickleweed exclude shorebirds and waterbirds from foraging or nesting on the mudflats. In drier upland areas, Indian fleabane (*Pluchea indica*), Brazilian pepper tree (*Schinus terebinthifolius*), and koa-haole *(Leucaena leucocephala*) form a dense thicket.

Red mangrove seeds first entered in the WMA in the early 1970s through culverts connecting the pond complex to adjoining bays. By 1974, the trees had become a pest species (Drigot 2000). Mangroves cover intertidal soft substrate in most of the tropics but are not native to Hawai'i. Red mangroves were first introduced to Hawai'i from Florida in 1902 to mitigate erosion after the destruction of coastal vegetation on the island of Moloka'i by humans and livestock (Merlin 1977). In 1922, 14,000 seedlings of red mangrove and three other mangrove species were planted in the saltmarshes of O'ahu. Within 50 years, red mangrove established a monotypic community in many fishponds, estuaries and sheltered coastlines in Hawai'i, estimated to be about 32% of all estuarine intertidal habitat in 1977 (Allen 1998). A similar situation occurred on Rodrigues Island (Indian Ocean) where unique mudflat habitat was destroyed by mangroves presumably planted to benefit wildlife (Sherley 2000).

Red mangrove grows as robustly in the Hawaiian coastal environment as in its native range. Odum (1970) found that red mangroves in south Florida shed their leaves at an annual rate of 9 metric tons per hectare (about 2.5 grams per m^2 per day). Studies from Nu'upia Ponds report 2.98 grams per m^2 per day (Cox and Jokiel 1996). Simberloff (1990) notes "the effect of this introduction on energy flow, nutrient cycling and succession must be enormous." Yet the relatively-recent introduction of mangroves (100 years ago) has not been long enough for many Hawaiian marine species to exploit the detritus-based food chain. Without a native mangrove ecological guild to benefit from increased productivity of mangrove habitat, native species give way to non-native species pre-adapted to mangroves.

In addition, mangrove propagules survive at a significantly higher rate in Hawai'i than they do in areas where indigenous seed predators exist (Steele *et al.* 1999). Mangroves quickly cover the wetland margins, which are an essential foraging habitat for key native Hawaiian wildlife species, and eventually displace the invasive pickleweed at the wetland's intertidal margins. Mangrove prop roots trap fine sediment and extend the shallow waters of fishpond edges – an undesirable condition in Hawai'i since it decreases water circulation, increases algal production, depletes dissolved oxygen levels, increases the temperature, acidity, and salinity levels, as well as accelerating deterioration of Nu'upia Ponds' historic fishpond walls.

The Nu'upia Ponds WMA is primarily managed by the Marines as a protected habitat for the federally endangered Hawaiian stilt (*Himantopus mexicanus knudseni*), an endemic subspecies of the black-necked stilt. Recent genetic and extant morphological and behavioural evidence suggests the insular Hawaiian stilt is a distinct species (Pratt and Pratt 2001). The stilt's optimal habitat is open mudflats with water depths of 13 cm or less and ponds of variable salinity (Engilis and Pratt 1993). Stilts using the WMA represent between 10% and 20% of the entire Hawaiian population that may fluctuate between 1200 and 1600 birds (Engilis and Pratt 1993).

Under military protection since World War II, Nu'upia Ponds became critical stilt habitat that aided their recovery from near-extinction. Habitat loss and hunting throughout Hawai'i reduced stilt numbers to about 200 birds statewide by the early 1940s (Munro 1944). A ban on hunting prior to World War II permitted the partial recovery of the population and a high of 128 stilts was recorded in 1948 at Nu'upia Ponds (Fig. 2). After 1948, stilt counts in the WMA unaccountably dropped; only two of 20 counts exceeded 50 birds from 1949 to 1964. There was also a period in late 1957 and early 1958 when, for unknown reasons, no birds were found. The average bird count from 1949 to 1964 was 30 birds. Stilt populations on O'ahu, including those at Nu'upia Ponds, have shown a steady increase coincident with active habitat management since the 1980s (Engilis and Pratt 1993).

METHODS

Systematic pickleweed control was begun in the early 1980s and has been crucial in maintaining open habitat for stilt feeding, loafing, and nesting. The vegetation is controlled annually during Marine Corps training with 26-ton Amphibious Assault Vehicles (AAVs) (Fig.3). These "mud ops" manoeuvres were initiated through collaborative consultation among MCBH environmental managers, state and federal wildlife biologists, and military operators, resulting in weed control and enhanced operator training under a variety of conditions (Drigot 2001). Pickleweed

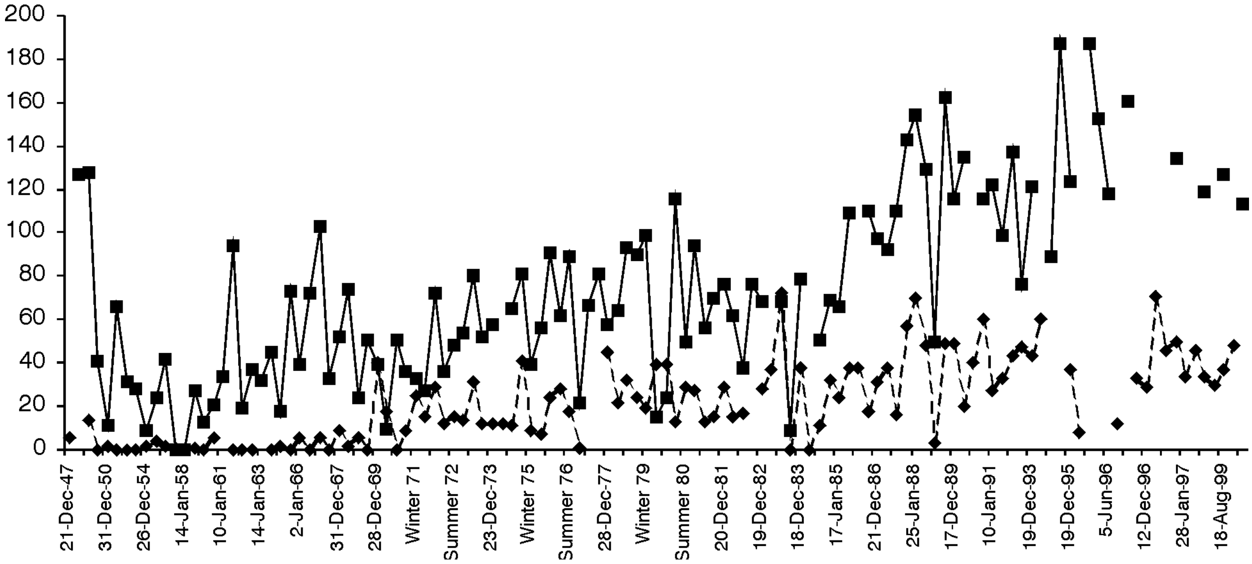

Fig. 2 Hawaiian stilt (squares) and black-crowned night-heron (diamonds) counts in Nu'upia Ponds. 1947-2001.

management consists of AAVs systematically plowing rows in the mudflats, much like a farmer plowing fields, creating "moat and island" terrain attractive to the ground-nesting stilt. The AAVs follow each other, with one set of tracks in the other's rut. Other AAVs crisscross perpendicular to these rows, resulting in a checkerboard mosaic pattern imprinted across the mudflats. In the process, the mud is churned up such that fine silt settles over the existing pickleweed rootstock, requiring the plants to grow from a new position. This recovery period may take from several months to years, during which time the Hawaiian stilts have a potential breeding habitat to exploit. Specific areas worked by AAVs may vary from year to year but plowing occurs annually in about 5 ha of the stilt breeding area.

In another portion of the area worked, AAVs run circular patterns to create "doughnut-like" patterns in the mudflats (Fig. 3). This landscape management activity provides stilt nesting islets surrounded by moats filled with water from groundwater seeps. The height and depth of the features vary but are generally less than a metre. These shallow, brackish to freshwater pools support dense stands of indigenous widgeon grass, aquatic insects such as shore flies (*Ephydra riparia)*, water boatmen (*Trichocorixa reticulata*) and fish such as non-native tilapia (*Oreochromis mossambicus*) and topminnows (*Poecilia mexicana*).

Fig. 3 Amphibious Assault Vehicle plowing "doughnuts" in pickleweed meadow.

In Nu'upia Ponds, at least 16 species of native fish find sanctuary but invasive tilapia comprise about a third of the total fish biomass (Brock 1995). The fry contribute to the stilt diet but the adult fish also consume resources that the stilts could eat. Stilts frequently follow AAVs during the "mud-ops" exercise to catch exposed small fish displaced by wave actions during plowing. It appears that stilts benefit from this action in the short term. However, after the manoeuvres, the mud dries and some of the exposed fish and invertebrates die. It appears to take several weeks before the AAV ruts fill with water from tides, rain, groundwater seepage, and the productivity recovers. Fallowing some mudflats in the stilt core-nesting areas for several breeding seasons may assist invertebrate species to reach maximum densities and provide improved feeding opportunities for stilt chicks. Fallowing also may allow pickleweed clumps to develop enough cover to protect stilt nests and young. However, excessive pickleweed provides cover for alien mammalian predators like rats, cats, and mongoose. Thus annual management is necessary.

Managed pickleweed appears to recover more slowly near the shore where poor drainage and high salinity impede its growth. However, this ecotone is where mangrove seedlings become established. Mangrove control in Hawai'i began in the early 1980s with volunteer labourers cutting mangroves growing in culverts and along trails (Devaney *et al.* 1982). In 1983, environmental managers at MCBH began to sponsor volunteer service projects to clear mangrove in the ponds. The intention was to deter further eastward expansion of this plant across the fishpond complex while awaiting sufficient funds for more large-scale removal of the well-established "seed-stock" of mature mangrove trees along the ponds' western flank (Drigot 2001). Volunteers, both military and civilian, used hand-held tools such as shovels and lopping shears. Only military personnel used chainsaws to help clear mangrove from historic fishpond walls. Large-scale mechanised mangrove removal began in the late 1980s when MCBH dredged the central storm-drain canal flowing through the central Mokapu drainage basin into Nu'upia Ponds and cleared mangrove in the ponds along a new nature trail. Through these experiences, it was discovered that young mangroves growing in water would not resprout if they were cut to the water line. If they grew above the high tide line and were sawed to not more than six inches above ground, they would not resprout. Only cut seedlings would coppice, so volunteer labourers, (e.g., various environmental, school and civic groups), periodically pulled seedlings. While only hand and mechanical removal methods have been used in Nu'upia Ponds, other mangrove-infested areas on O'ahu have been successfully treated with Garlon 4™ basal treatment.

With receipt of several large federal grants from Headquarters Marine Corps and from the Department of Defense Legacy Program, eradication of the remaining mangrove using heavy equipment became a more feasible goal. In 1995, after completion of an Environmental Assessment, a lengthy permitting process with U.S Army Corp of Engineers, State of Hawai'i Department of Health and consultations with state and federal wildlife and historic preservation agencies, the removal began. By avoiding stilt nesting season, and using appropriate archaeological monitoring near fishpond walls, silt containment booms and water quality monitoring along coastline affected areas, mature mangrove stands were removed using tracked heavy equipment, a Catel 200™ with a grappling arm. (Fig. 4). Mangrove debris was chipped in a tub-grinder and deposited along the surface of pond access roads. This practice avoided the added expense of hauling chips to a landfill. By 1999, virtually the full extent of mangroves within the Ponds interior had been removed (an estimated 20 acres) at a cost approximating USD 2.5 million dollars (Drigot 2000). Marine Corps environmental managers are now focused on clearing additional mangroves along MCBH shorelines facing Kane'ohe Bay outside the pond perimeter. Mangrove infestation in the Bay provides a "seed bank" for re-entry into the ponds through culverts that allow water exchange.

Fig. 4 Mangrove removal using heavy equipment, tub grinder and grappling arm.

In 1994 and 1996, before and during the peak of mangrove eradication and directed pickleweed management, the Marine Corps funded studies of the reproductive biology of the Hawaiian stilt during the breeding seasons (Rauzon and Tanino 1995). Censuses of all waterbirds were conducted bi-monthly during the breeding study and compared with those performed bi-annually over the span of a half-century by state and federal biologists, using spotting scopes and binoculars.

Observer bias varied over this lengthy period, due in part to variations in individual effort and time of day surveyed. By the mid-1980s, mangroves had grown up and obscured much of the viewshed so counts were limited to open mudflat areas. Night-herons roosting at midday in dense mangroves in inaccessible areas were easily overlooked while stilts remained conspicuous on the open mudflats. Since 1996, only semi-annual population counts were consistently made and stilt reproductive output can only be inferred from counts of fledged chicks.

RESULTS

Hawaiian Stilts

Figure 2 portrays 52 years of stilt surveys, conducted during the bi-annual statewide waterbird counts, the Audubon Christmas bird counts, and various researcher censuses. While highly variable over time, Hawaiian stilt numbers in the WMA increased after the implementation of pickleweed management in the early 1980s. From 1965-1975, the period before management, an average of 54 birds/count was obtained with a high of 103 birds. From 1976-1980, counts averaged 88 birds/count, with a high count of 124. From 1981-1985, the average was 66 birds/count and count lows were less than 40 birds. (Table 1).

Since 1983, management actions such as regular AAV pickleweed plowing manoeuvres with intensified mammalian predator trapping efforts, and minimised human disturbance, coincided with significantly-higher bird counts (Drigot 2000). In 1987, stilt counts at the Base exceeded the earliest, highest counts during 1947 and 1948, (127 and 128 birds respectively). In 1989, 169 stilts were recorded in the July bi-annual waterbird census. By the mid-1990s, during intensive habitat management activities, stilt numbers climbed to the highest average counts ever recorded in the WMA. The mean number of stilts in the WMA increased from 129 in 1994 to 145 in 1995 and 135 in 1996, with the highest count, 187, recorded in 1995 (Rauzon and Tanino 1995; Rauzon *et al*. 1997) (Table 1).

Mangrove removal at Nu'upia Ponds WMA also had an immediate and positive effect on Hawaiian stilts. Only a few stilt were seen using the mangrove-infested peninsula within Nu'upia 'Ekahi Pond in 1994, but by February 1995,

Table 1 Hawaiian Stilt Census in Nu'upia Ponds: 1978-2001

Years	Mean	Counts	Range	S.D.
1976-80	88	15	50-124	24.2
1981-85	66	14	38-109	17.8
1986-90	117	12	50-162	30.2
1991-93	106	12	75-137	19.2
1994	129	41	89-162	15.2
1995	146	18	124-187	18.2
1996	135	21	118-164	14.2
1997	129	7	107-161	20.2
1998	129	2	119-139	14.1
1999	122	2	116-127	7.8
2000	113	1	113	0
2001	129	2	112-146	0

with the near-complete clearance of mangroves, numerous stilts began using areas from which they were previously excluded. In 1995, a nest was placed in this area, judging by observed stilt defensive behaviour. In March 1994, stilt nests were made in newly-cleared mangrove islets. One nest was lost to flooding but another produced several chicks. At least three other nests were located along the newly-cleared pond margins. Heleloa Pond, newly cleared of mangroves, had two pairs of stilts move in, even while heavy equipment operated nearby.

Observed nesting attempts (n=8), including repeat nesting, abandoned and "dump" nests, increased 50% between 1994 and 1996. This was likely due to increased habitat available (e.g., through intensified mangrove removal and pickleweed management) and increased observer experience in finding cryptic nests. However, despite increases in the number of nests, eggs laid, and chicks hatched, there appeared little increase in the number of overall fledglings produced within Nu'upia Ponds (Table 2). In 1994, at least 191 eggs were laid and about 23 chicks fledged. In 1996, at least 297 eggs were laid, yet a similar number of chicks fledged as in 1994. Hatching success improved (1994 = 0.24: 1996 = 0.72), while fledgling success declined (1994 = 0.51: 1996 = 0.12).

Black-crowned Night-herons

The indigenous black-crowned night-heron (*Nycticorax nycticorax hoatil*) is resident in the WMA, primarily feeding on tilapia and nesting in mangroves and other introduced trees. The night-heron is the only native waterbird in Hawai'i (stilts, coots, moorhens, and ducks) that is not an endemic species or subspecies. Because it has not genetically differentiated from stock on the American continent, it is not a federally protected migratory species. It is a state-protected species but permits are obtainable for lethal control of local populations when, for example, they cause significant depredation in mariculture areas. In fact, statewide increases in night-herons appear linked to mariculture expansion (Engilis and Pratt 1993).

Night-herons in Nu'upia Ponds have benefited from both pickleweed management and mangrove maturation. Pickleweed clearing opens up foraging habitat while dense mature mangrove thickets are critical for night-heron nesting by providing isolation from potential predators and human disturbance. In the WMA, before vegetation control efforts began and while mangroves were short, heron counts were usually less than 10 birds. By 1995, the average count was 36 (n=41, range 14-72) (Rauzon and Tanino 1995).

Table 2 Hawaiian Stilt Reproductive Success: 1976-2001

Year	Nests	Eggs	Chicks	Fledglings	# Count	Range	S.D.	H.S[2]	F.S [3]
1978	24	86	?	?				0.23	
1979	31	102[1]	49	26	?			0.48	0.53
1980	43	139[1]	72	8	?			0.52	0.11
1981	29[1]	?	?	13	?				
1982	?	?	?	17	?			0.34	
1994	56	191	45	23.0	14	16-30	3.9	0.24	0.51
1995	?	?	?	25.1	14	15-33	5.4		
1996	84	297	215	25.5	10	17-35	7.5	0.72	0.12
1997	?	?	?	26	2	25-27			
1998	?	?	?	?	?				
1999	?	?	?	21	1				
2000	?	?	?	?					
2001	?	?	?	6	1				

[1] inferred from data.
[2] H.S.-Hatching success is the ratio of the eggs hatched to eggs laid. The data is based on nests located during searches but may not represent the entire stilt reproductive effort.
[3] F.S. -Fledging success is based on the ratio of the total number fledged to the total number of chicks hatched, and is derived from a mean calculated from observations of fledglings beginning in July through to the end of December.

Figure 2 shows the censuses of black-crowned night-herons over a 52-year period. Their average population is about a third of the Hawaiian stilt population (36 verses 122, respective means) and like stilts, night-heron counts vary over time, due in part to vegetation, observer effort and time of day. In the mid-1980s, mangroves obscured much of the viewshed so counts were limited to open mudflat areas. Night-herons roosting at midday in dense mangrove were easily overlooked. In fact, night-heron nests were only first discovered in the WMA during mangrove removal in 1994, although they probably nested for years without detection. Their stick nests were placed approximately eight to 20 feet off the ground in large mangroves.

During the mangrove removal process, night-herons were discovered nesting in trees scheduled to be cut in 1996-97. In the summer of 1994, 39 nests were located, meaning that at least 78 adults were present and with 15% of the population being juveniles, approximately 100 birds were resident in the WMA (Rauzon and Tanino 1995). During field surveys prior to cutting, on 16 December 1996, we found at least 23 night-heron nests, representing 46 adult birds. After meeting all legal requirements from the federal and state regulators, 31 night-heron nests were eventually destroyed in Nu'upia Ponds. In order to salvage some scientific data from the operation, eggs were measured to yield a mean length of 50.97 mm, and width 36.49 mm (n=42). The eggs of two night-heron nests were collected and donated to the Bishop Museum in Honolulu in compliance with permit conditions.

Two nests with chicks were saved from destruction. The trees surrounding the nest were flagged with pink tape to alert the cutters. One nest held one chick and two eggs on 17 December 1996. By the next day, the second egg hatched and the third egg pipped. By subtracting the approximate 30-day incubation length, the eggs were laid in mid-to-late November. When this nest was revisited on 31 January 1997, there was one dead chick in the nest. The others fledged or died earlier and disappeared. An immature fledgling and another dead chick were later seen in the same nest in August 1997.

A second occupied nest had 3 eggs on 17 December 1996 and again on 31 January 1997. Re-nesting had occurred since the incubation period does not extend to 45 days. The second clutch hatched around 7 February 1997. The chick with emerging pinfeathers was still alive on 27 February 1997, and presumably fledged in spite of the disturbance from nearby chainsaw activity. This nest site also held a large juvenile in August. These observations suggest that several pairs use nest sites throughout the year, perhaps successively. Night-herons continue to breed in other mangrove-infested areas off Base, and in Brazilian pepper trees and kiawe (*Prosopis pallida*) fringing the WMA. They also continue to forage at Nu'upia Ponds, often feeding in the stilt nesting area.

Feeding night herons were counted during each stilt survey from March 1994 until February 1995. The range of 41 counts spanned 14 to 72 with the mean being 36.4 birds. Six counts made during late 1996-early 1997 yielded a mean of 41 herons. In 2000, 48 night-herons were counted, suggesting their population and use of the ponds is consistent in spite of nesting habitat loss. However, it is very likely populations would have increased if mangroves remained.

Night-herons exert an unknown but possibly significant predation pressure on stilt eggs and chicks. Although we saw no direct evidence of predation at Nu'upia Ponds, "All available evidence points to black-crowned night-herons being extremely opportunistic predators utilising whatever suitable prey happens to be most plentiful or most easily caught at any particular place and time." (Collins 1979). Wolford and Boag (1971) found night-herons in Alberta, Canada, fed on blackbird, egret, ibis, duck, gull, and tern chicks. Shallenberger (1977) found regurgitated pellets containing a sooty tern (*Sterna fuscata*) chick under a Hawaiian night-heron roost and an adult night-heron was observed eating a stilt chick at James Campbell National Wildlife Refuge, O'ahu (Andrews 1981).

Cattle Egrets

The cattle egret (*Bubulcus ibis*) is an Old World species that dispersed across the Southern Atlantic Ocean to become established in South America in the 1940s. The birds moved north and west and reached Florida in 1948. In 1952, they colonised Canada and Bermuda on their own (Matthiessen 1959). Cattle egrets were introduced from Florida to Hawai'i in 1959. About 105 individuals were released to control sugarcane-eating insects and flies that pester cattle (Breese 1959).

Cattle egrets are considered to be a pest species in Hawai'i. They prey on chicks of the endangered stilt and Hawaiian coot (*Fulica alai*) (Andrews 1981), and potentially carry diseases (*Salmonella*) that might spread to other birds. Egrets are also a threat to aircraft because birds forage in grass strips near runways and increase the bird/aircraft strike hazard potential. Several airfields exercise lethal control under the authority of state and federal permits.

The first cattle egret roost was discovered in the WMA in the 1960s. About 30 nests were active in kiawe trees on 5 October 1970 (Olsen 1970). The colony expanded annually, and by 1977, the roost was described as the largest in Hawai'i (Shallenberger 1977). Christmas Count totals of roosting birds from 1976 through 1979 were consistently within 200 birds of the mean, 1105 birds. In the spring of 1982, the nesting colony moved to another tree in Nu'upia Pond. About 175-200 birds used this site until January 1983 when a new roost formed in mangroves at He'eia Fishpond, 3.2 km west of MCBH. The egrets abandoned the Nu'upia Ponds colony but continued to forage in lawns and other grassy areas on Base.

During a lull in mangrove removal, cattle egrets began nesting in a mangrove islet in Nu'upia Ponds. They were attracted to this site because of the size, isolation, and wind protection of the island. The birds began roosting in March

1996 and were breeding by June. Nests were again detected in November 1996. Incubation is 22-26 days with fledging in about 30 days, so eggs would have hatched in December-January period with chicks fledging in February to March; around the stilt nesting season.

Cattle egrets breed year-round in the tropics with different regional peaks (del Hoyo *et al.* 1992). Paton *et al.* (1986) found that there was no period between January and June that egrets did not nest in Hawai'i. The nesting island was scheduled to be cut in the winter, presumably when fewer birds would be breeding. Immediately before the scheduled mangrove cut, and with proper state and federal permits for hazing birds, 195 cattle egret nests were destroyed with a long pole. We measured 185 cattle egret eggs. Egg size appears to be the same as "normal", with a 45.55 mm length and 32.82 mm width (Telfair 1983). Paton *et al.* (1986) reported a mean clutch size of 3.32 eggs (n=41, s.d.=1.06), but we cannot provide comparable data since the nests were destroyed during nest initiation.

After the mangrove colony was destroyed, the bulk of the birds presumably returned to the large colony at He'eia Fishpond, also on a mangrove islet. Some individuals continued to roost nearby the former site in the remaining mangrove until these were cut. Cattle egrets continue to forage on MCBH, but now have to "commute" 3.2 km to roost.

Great Blue Herons

One great blue heron arrived as a vagrant to the WMA in late 1995 and two subadult herons arrived in early April 1996. One bird, probably an adult, subsequently disappeared and the two juveniles remained in the WMA to early 1997 in spite of much human activity during mangrove clearing. The two herons roosted in the cattle egret rookery until it was cut down.

Great blue herons have "wandered a few times to the Hawaiian Islands" (Pratt 1987). The previous record from Nu'upia Ponds is of an individual in the early 1980s. In 1996, several were recorded around the Hawaiian Islands, and it appears they all ended up at Nu'upia Ponds at least for a short time. This demonstrates that there are few areas capable of supporting great blue herons, and Nu'upia Ponds WMA, while infested with mangrove, was probably one of the best sites for them in the State because of abundant cover and food.

If the herons were of opposite sex and eventually bred in the WMA, this would have been a rare opportunity to detail a North American bird colonising Hawai'i in historic times. Like night-herons which arrived unaided by man and colonised relatively recently, and given the prevalence of other mangrove-infested wetlands that remain on O'ahu, great blue herons may yet become established in an alien-dominated landscape in Hawai'i. However, due to legally-driven priorities to restore endangered species habitat and a historic Hawaiian landscape, this opportunity was foregone.

DISCUSSION

Since 1980, the number of Hawaiian stilts observed on biannual counts has almost doubled in Nu'upia Ponds to include at one time up to 20% of the estimated total Hawai'i population (Rauzon *et al.* 1997). This increase coincides with intensified vegetation management; pickleweed control with AAVs, large-scale mangrove removal, an active predator control programme, clearing clogged culverts, and restricting human access. In recognition of environmental stewardship, the Base command has received multiple national, state and local awards (Drigot 2000).

State-wide stilt numbers also increased 114%, at the same time Nu'upia numbers were rising. Engilis and Pratt (1993) suggest that Hawaiian stilt populations appear inversely related to rainfall. Population increases occurred during a Southern Oscillation (El Niño) pattern of drier-than-normal Hawaiian weather from 1983-1985 and 1994-1995 (Haraguchi and Matsunaga 1985; Engilis and Pratt 1993). Excessive rainfall during the breeding season can cause nest flooding and increased mortality to stilt chicks while winter rains enable stilts to exploit seasonal foraging habitats (Meininger 1990). Drought years can expose more mudflats and create new islands.

Separating the direct cause and effect of vegetation management programmes when large-scale climatic actions (e.g., El Niño) are in effect, is difficult. However, we have direct evidence of the positive influences vegetation management has on stilt populations. The discovery of stilt nests in mangrove stubble on areas cleared of mangroves demonstrates that stilts are flexible in nesting choices and that they will quickly exploit new areas that are near established breeding areas. As stilts may be approaching maximum nesting densities in historic breeding areas of the ponds, young birds' first attempts to nest may be in these adjacent mangrove-cleared areas.

During the past 30 years, establishment of red mangrove has facilitated egret and heron use of the ponds and increased the threat of predation on stilts. With maturation of mangroves, cattle egrets, which normally do not forage in saltmarshes, became common in the ponds, and an active rookery was established. With the mangrove infestation and cattle egret colony, these Hawaiian fishponds began to take on the character of a southern Florida landscape. Even the presence of alien pickleweed and tilapia added to the south Florida ecosystem, since both species are common and successful introductions to Florida as well.

Following the principles of ecosystem management, any special interest in preserving a transplanted Florida environment or in attending to the needs of one or more species of special interest (e.g., night-herons or great blue herons) must succumb to the paramount objective to "maintain and improve the sustainability and native biodiversity of ecosystems" and base resource management decisions on "best science" and "associated cultural values" (Drigot 2001). Nu'upia Ponds is a recognised national historic property for its ancient Hawaiian fishpond/landscape characteristics, so preserving this cultural landscape devoid of mangrove takes precedence over any special-interest concerns about preserving an invasive mangrove forest despite threats to mangrove forests elsewhere in the world.

Furthermore, unlike red mangrove and great blue herons, endemic endangered stilts are found nowhere else in the world and clearly benefit by removal of potential predators, such as all three Ardeidae, which have been observed eating small birds, and would take stilt chicks as well. Management priorities of maintaining open mudflats for endangered Hawaiian stilts precluded any habitat management for indigenous and alien waterbirds. Also, mangrove impacts to the native Hawaiian fishpond walls, water quality changes due to decreased circulation, and clogged channels preventing fish movement, adversely affected the health of the fishpond ecosystem.

Since the bulk of the mangrove was removed in 1997, water circulation and dissolved oxygen levels have increased (Drigot 1999). Stilt populations have dipped and counts of young produced after mangrove removal have not sustained a population boost due to new habitat availability. This is likely due to pickleweed quickly colonising newly-cleared areas. Stilts nesting in disturbed areas one year may not have that opportunity to nest in the subsequent year without additional vegetation management. Other factors that may play a role in affecting the decline may be nest flooding, limited food choice due to tilapia competition, predation by alien mammals and dispersal.

In contrast to Hawaiian stilts, fecundity of cattle egrets is especially high. Within one year of establishing a colony, egrets produced over 200 nests with about 500 eggs, clearly demonstrating why the species has undergone such enormous global expansion. One reason for the birds' success is its unique ability among Ardeidae to breed when they are one year old (Kohlar 1966). One clutch per year is usually laid but up to three has been recorded, with usually 2-6 eggs per clutch (Berger 1981). This fecundity is coupled with behavioural adaptability to anthropogenic disturbances and abundant food. By feeding on introduced species (cockroaches, centipedes, mice, etc.) that are exposed by large grazing ungulates and lawn-mowing machinery, egrets fill an unoccupied ecological niche in Hawai'i as elsewhere. Cattle egrets benefit the Base by eating many introduced pests. However, they pose a hazard to aircraft safety due to their propensity to forage in grassy airfield borders, and are reputed to carry avian diseases that could spread to native bird populations.

No native Hawaiian species have yet learned to adjust their behaviours so precisely to human's ways, although the Hawaiian stilt may owe its survival to being able to forage on introduced food items as well. For example, stilts are commonly observed foraging on cockroaches in grassy inland areas of the Base, and forage daily at the polishing ponds at the Base water reclamation facility.

Mangroves are essentially eradicated from the Nu'upia Ponds WMA, although seeds float into the ponds from the outer bays where mangroves remain uncontrolled in coastal areas outside MCBH jurisdiction. Sprouted propagules must be pulled up on a continual basis by volunteer service groups until an effective seed filter is in place at inflow channels. Future mangrove management may lie in biocontrol. The first steps in exploring a biocontrol strategy, albeit using another alien species, is underway. Mangrove propagules were sent from Hawai'i to Louisiana to test them for susceptibility to a beetle (*Poecilips/ Coccotrypes rhizophorae*) that reduces the production of viable seeds (Allen 1998).

A promising tool for regional mangrove management is a specialised amphibious excavator, recently purchased by the City and County of Honolulu in partnership with Ducks Unlimited and the U.S. Fish and Wildlife Service. Other state, federal and local landowners with similar wetland management responsibilities are evaluating possible ways to leverage their individually-limited resources through cooperative use of this equipment on alien species whose spread remains indifferent to jurisdictional boundaries. With the advent of such interagency partnerships to the arsenal of alien species management tools, it is hoped that one day soon the Hawaiian stilt may be removed from the Endangered Species list.

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When is eradication of exotic pest plants a realistic goal?

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Abstract Using a unique data set on eradication attempts by the California Department of Food and Agriculture on 18 species and 53 separate infestations targeted for eradication in the period 1972-2000, we show that professional eradication of exotic weed infestations smaller than one hectare is usually possible. In addition, about 1/3 of infestations between 1 ha and 100 ha and 1/4 of infestations between 101 and 1000 ha have been eradicated. However, costs of eradication projects increase dramatically. With a realistic amount of resources, it is very unlikely that infestations larger than 1000 ha can be eradicated. Early detection of the presence of an invasive taxon can make the difference between being able to employ offensive strategies (eradication), and the necessity of retreating to a defensive strategy that usually means an infinite financial commitment. Nevertheless, depending on the potential impact of individual weedy species, even infestations larger than 1000 hectares should be targeted for eradication effort or, at least, substantial reduction and containment. If an exotic weed is already widespread, then species-specific biological control may be the only long-term effective method able to suppress its abundance over large areas.

Keywords Costs of eradication; early detection; eradication effort; exotic pests; initial infestation; invasive plants; noxious weeds.

INTRODUCTION

Many control methods and their combinations (usually involving mechanical, chemical, and biological means) are available to managers for containing, controlling, or eradicating harmful alien plants. However, sound management strategies demand an objective means for setting priorities. Undoubtedly, exotic taxa with large-scale environmental impacts ("transformers" – see Richardson *et al.* 2000; Rejmánek *et al.* 2002) should always be targets for control and eradication. But when is complete eradication a realistic goal? There are numerous examples where small infestations of invasive plant species have been eradicated. These include *Silybum marianum* on Santa Barbara Island and *Osteospermum fruticosum* on Santa Cruz Island, California (Junak *et al.* 1993; Junak pers. comm.), *Pueraria phaseoloides* in Galápagos (Soria *et al.* 2002), and nine species on Rangitoto Island (Wotherspoon and Wotherspoon 2002) . There are also several encouraging examples where widespread alien animals have been completely eradicated (Dahlsten and Garcia 1989; Chapuis and Barnaud 1995; Priddel *et al*. 2000; more examples are in this volume). Can equally widespread and difficult alien plants also be eradicated? We try to answer this question by using a unique data set on exotic weed eradication attempts by the California Department of Food and Agriculture.

The California Department of Food and Agriculture (CDFA) is actively involved in preventing the establishment and invasion of "noxious weeds." The Food and Agricultural Code of California defines a noxious weed as "any plant species which is, or is liable to be, detrimental or destructive to agriculture, silviculture, or important native species, and difficult to control or eradicate." Each noxious weed is given a pest rating (A, B, C, or Q) which indicates the most appropriate action to be taken against it (O'Connell 1999). An "A" rated weed is subject to action by the CDFA and County Agricultural Commissioner Offices including eradication, quarantine, containment, rejection of shipments, or other holding actions. A "B" rated weed is subject to State action only when found in a nursery; otherwise action is at the discretion of the local County Agricultural Commissioner. A "C" rated weed is not subject to State action other than to provide for general cleanliness in nurseries, otherwise action is at the discretion of the local County Agricultural Commissioner. Those weeds that are widespread and can no longer be eradicated are usually given a "C" rating. A weed is rated "Q" when it is newly detected and seems likely to significantly impact agriculture. These weeds are treated as "A" rated until they are fully evaluated. Currently, there are 128 plant species that are listed as "noxious" by CDFA: 45 are "A" rated, 55 are "B" rated, 24 are "C" rated, and 4 are "Q" rated.

Eradication and other actions directed at "A" rated weeds are performed by personnel in the Integrated Pest Control Branch of CDFA and the County Agricultural Commissioner Offices who work closely together to detect and eradicate exotic weeds state-wide. When a new infestation of an "A" rated weed is detected, the site is visited and size of the infestation is delimited. Two estimates of infestation size, net and gross, are obtained. Gross infestation size is the area over which the weed is distributed. Net infestation size is the area to which treatment is actually applied. Gross infestation size is the area that must be surveyed in return trips following control treatments.

Eradication efforts consist of a series of control treatments to the infestation over several years. Control treatments can include herbicide applications, cultivation, removal of infested soil, and mechanical removal. For large infestations, a crew of workers is required; for small infesta-

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tions, only one individual may complete the work. Following initial treatment, the site is visited several times to examine the area for regrowth or seedling recruitment. This effort is repeated until no plants are found in subsequent visits. Eradication is considered successful when no plants are recovered from the initial infested area for three consecutive years.

To date, 14 exotic weeds have been successfully eradicated from California: whitestem distaff thistle (*Carthamus leucocaulos*), dudaim melon (*Cucumis melo* var. *dudaim*), giant dodder (*Cuscuta reflexa*), serrate spurge (*Euphorbia serrata*), Russian salttree (*Halimodendron halodendron*), blueweed (*Helianthus ciliaris*), tanglehead (*Heteropogon contortus*), creeping mesquite (*Prosopis strombulifera*), heartleaf nightshade (*Solanum cardiophyllum*), Torrey's nightshade (*Solanum dimidiatum*), Austrian peaweed (*Sphaerophysa salsula*), wild marigold (*Tagetes minuta*), Syrian beancaper (*Zygophyllum fabago*), and meadowsage (*Salvia virgata*) (O'Connell 1999). With the exception of *Cucumis* (16 and 32 ha), all gross infestations were smaller than 10 ha and most of them were smaller than one hectare when they were detected.

MATERIAL AND METHODS

Complete information on eradication effort was obtained for 53 infestations of 18 "A" rated species (Table 1). CDFA biologists assigned to the Detection and Eradication Districts for the State of California, CDFA, provided the data. For each weed infestation, the following information was obtained: (1) size of infestation after delimitation (both net and gross area), (2) date first found, (3) total number of visits to the site to date, (4) effort per infestation (number of person hours devoted to the site to date, including travel time to and from the site), and (5) current status of the infestation. The data are summarised in this contribution.

RESULTS

The relation between the mean eradication effort (work hours) and five initial gross infestation area categories is summarised in Table 2 and Fig. 1. The good news is that professional eradication of exotic weed infestations smaller than one hectare is usually possible. Furthermore, about 1/3 of all infestations between 1 ha and 100 ha and 1/4 of infestations between 101 and 1000 ha have been eradicated. Costs, however, increase dramatically. (An approximate estimate of direct costs in USD can be obtained by multiplying work hours in Fig. 1 and Table 2 by USD96; this includes salaries, cost of transportation, and cost of herbicides and equipment). With a realistic amount of resources, it is very unlikely that infestations larger than 1000 ha can be eradicated.

Interestingly, in the first four infestation-size categories, where at least some eradications were successful (Table 2), mean eradication effort per infestation is consistently greater for ongoing projects than for eradicated infestations. This indicates that, in general, completed eradications were not successful because of the greater effort.

Table 1 List of "A" rated weeds in California for which eradication information was obtained.

Scientific name	Common name	No. infestations	Eradicated/ongoing
Terrestrial species			
Alhagi pseudalhagi	camelthorn	5	1/4
Carduus nutans	musk thistle	1	0/1
Centaurea diffusa	diffuse knapweed	6	5/1
Centaurea iberica	Iberian thistle	3	1/2
Centaurea maculosa	spotted knapweed	3	2/1
Cirsium ochrocentrum	yellowspine thistle	3	1/2
Cucumis melo var. *dudaim*	dudaim melon	1	1/0
Cuscuta reflexa	giant dodder	1	1/0
Euphorbia esula	leafy spurge	2	1/1
Halimodendron halodendron	Russian salt tree	1	1/0
Linaria angustifolia ssp. *dalmatica*	Dalmatian toadflax	1	1/0
Onopordum acanthium	Scotch thistle	13	6/7
Onopordum illyricum	Illyrian thistle	1	0/1
Peganum harmala	harmel	2	0/2
Physalis viscosa	ground cherry	1	1/0
Salsola damascena	Damascus saltwort	1	0/1
Aquatic species			
Hydrilla verticilata	hydrilla	5	2/3
Alternanthera philoxeroides	alligatorweed	3	1/2

Another confounding factor could be a bias created by differences in species representing small and large infestations. This would be particularly serious if large infestations consisted of more persistent species than smaller infestations. However, the trend remains the same even within individual species (Fig. 2). Finally, while the eradication effort increases with the area of infestation, the effort per hectare decreases at the same time (Table 2). This suggests that even infestations of >1000 ha could be eradicated, but the eradication effort per hectare would have to be greater. It is important to point out that all three successful eradications of gross infestations >100 ha (Table 2) represented relatively-small net areas (*Linaria angustifolia*: 0.49 ha; *Onopordum acanthium*: 0.20 ha; *Physalis viscosa*: 0.92 ha).

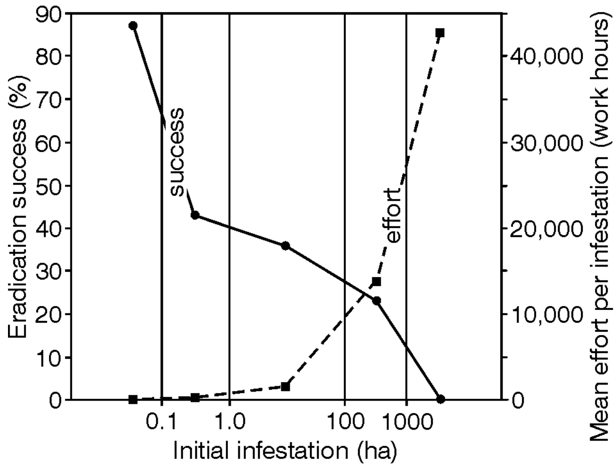

Fig. 1 The dependence of the eradication success (%) and the mean eradication effort per infestation (work hours) on the initial size of infestations. Based on the data for eradication projects of 18 noxious weed species and 53 independent infestations in California (see Table 1).

DISCUSSION

Obviously, a substantial increase in resources for exclusion and early detection of exotic weeds would be the most profitable investment. Without any data, or based on very limited data, others (Auld *et al.* 1987; Chippendale cited in Hobbs and Humphries 1995; Cook and Setterfield 1996; Braithwaite and Timmins 1999; Panetta 1999; Smith *et al.* 1999; Weiss 1999) already made this point. Surprisingly, however, practical implementations are still very rare. We suggest that in all concerned countries, teams of professional botanists should be created for rapid detection and assessment of new infestations of exotic plants. Early detection of the presence of an invasive and harmful taxon can make the difference between being able to employ feasible offensive strategies (eradication) and the necessity of retreating to a defensive strategy that usually means an infinite financial commitment.

Attempts to eradicate widespread invasive species, especially those that do not have any obvious environmental impacts (including suppression of rare native taxa), may be not only hopeless but also a waste of time and resources (Groening and Wolschke-Bulmahn 1992). Volunteers and donors, who would be otherwise willing to participate in eradication of serious pests, may be discouraged by such projects.

Nevertheless, depending on the potential impact of individual weedy species, even infestations larger than 1000 hectares should be targeted for eradication effort, or, at least, substantial reduction and containment. A notable example of a successful containment is the parasitic weed *Striga asiatica* in parts of North and South Carolina (Kaiser 1999). In the 45 years of the eradication programme, the initial gross infestation on 20 000 km^2 was reduced to 2800 ha of very light occurrences. The cost, however, was more than USD 250 million (R. E. Eplee, pers. comm.). Another exceptionally successful project is the practically complete eradication (98% of properties on which it is known to occur) of *Bassia* (*Kochia*) *scoparia* over the past eight years in Australia (3277 ha; 15,536 work hours; R. Randall, pers. comm.).

Table 2 Areas of initial gross infestations (at the beginning of eradication projects) of exotic weeds in California, numbers of eradicated infestations, numbers of ongoing projects, and mean eradication effort for five infestation area categories. The data include 18 species of noxious weedy species (two aquatic and 16 terrestrial) representing 53 separate infestations. NA – not applicable.

		Initial infestation (ha)				
		<0.1	0.1-1	1.1-100	101-1000	>1000
No. of eradicated infestations		13	3	5	3	0
No. of ongoing projects		2	4	9	10	4
Mean eradication effort per infestation (work hours)	Eradicated	63	180	1496	1845	-
	Ongoing	174	277	1577	17 194	42 751
Mean eradication effort per hectare (work hours)	Eradicated	NA	807	103	6	-
	Ongoing	NA	792	648	26	16

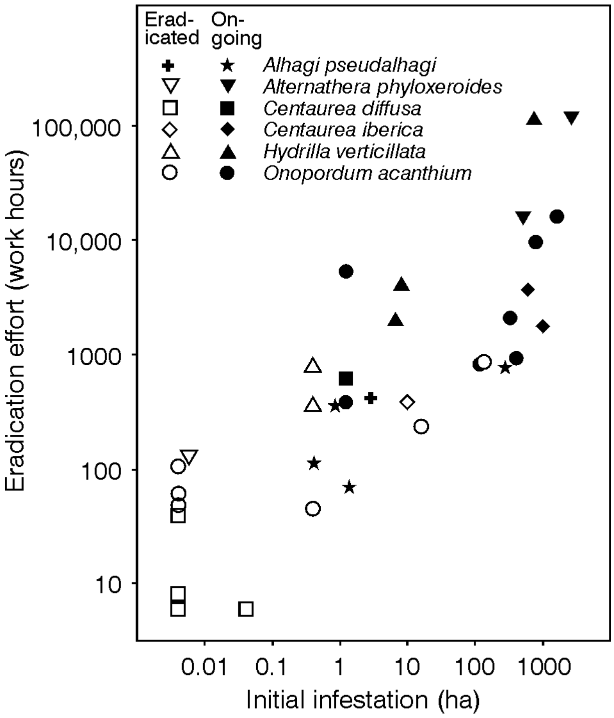

Fig. 2 The dependence of eradication effort on the size of initial infestations of six noxious weedy species in California. This diagram also indicates that eradication of aquatic weeds (*Alternathera* and *Hydrilla*) is more demanding.

In general, however, when an exotic weed is already widespread (>10 000 ha), species-specific biological control (if feasible) may be the only long-term effective way to suppress its abundance over the invaded area. Many successful weed biocontrol projects have been accomplished in Australia, California, South Africa, and other countries (Nechols 1995; Julien and Griffiths 1998; Olckers and Hill 1999; Pemberton 2000). Needless to say that as biological control agents are usually exotic taxa themselves, serious attention must be paid to their possible non-target effects (Louda *et al.* 1997; Wajnberg *et al.* 2001).

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Management of indigenous and alien Malvaceae on islands near Perth, Western Australia

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Abstract *Malva dendromorpha,* European tree mallow, has not previously been described as a serious environmental weed. Over the last two to three decades tree mallow has invaded seabird nesting islands off the West Australian coast, growing in dense clumps to 3m tall and outcompeting native perennial species including *Malva australiana*, the Australian native hollyhock. As a result of excessive growth of tree mallow on small islands there has been a serious loss of biodiversity. The stands of biennial tree mallow have no understorey and die back in the hot, dry summer, exposing the soil to erosion by the strong sea breezes and also rendering it vulnerable to invasion by annual weeds. The habitat may then be less suitable for nesting bird colonies. Control measures on Seal Island over two years included regular mechanical removal and stump treatment with Roundup (glyphosate). Subsequently weedmat was laid down and some native species were planted. While eradication was not possible, control produced an ecologically-desirable outcome. There has been a 70% reduction in the cover of tree mallow and the native hollyhock has re-established itself locally. Other planted native species failed to survive amidst heavy growth of annual invading alien grasses and herbs which included *Malva parviflora* (marshmallow). Similar invasions by tree mallow have occurred on islands in South Australia and Victoria. In South Australia, management options were investigated but adequate resources to put them into practice were not available. In Victoria, regular hand removal of tree mallow over a seven year period has virtually eliminated tree mallow and the native hollyhock is flourishing. The problems we encountered are summarised and future directions outlined.

Keywords Tree mallow, *Malva dendromorpha*; native hollyhock, *Malva australiana*; island vegetation, weed management.

INTRODUCTION

The islands

In 1997 we surveyed the vegetation of four islands in the Shoalwater Islands Marine Park, some 40 km south of Perth, Western Australia: Penguin Island (12.5 ha), Middle Shag Island (0.4 ha), Seal Island (1.2 ha) and Bird Island (0.9 ha) (Rippey *et al.* 1998) (Fig. 1).

These islands are composed of aeolianite limestone, residuals of an old dune system, which was inundated some 6000 years ago when sea levels rose (Playford 1988). The islands were isolated and now form part of the present parallel limestone reef system. A variable layer of calcareous sand overlies the limestone forming beaches and dunes.

On Penguin Island public access is restricted to the beaches, a picnic area, and boardwalks that cross the island in two places. No landing is permitted on the smaller islands.

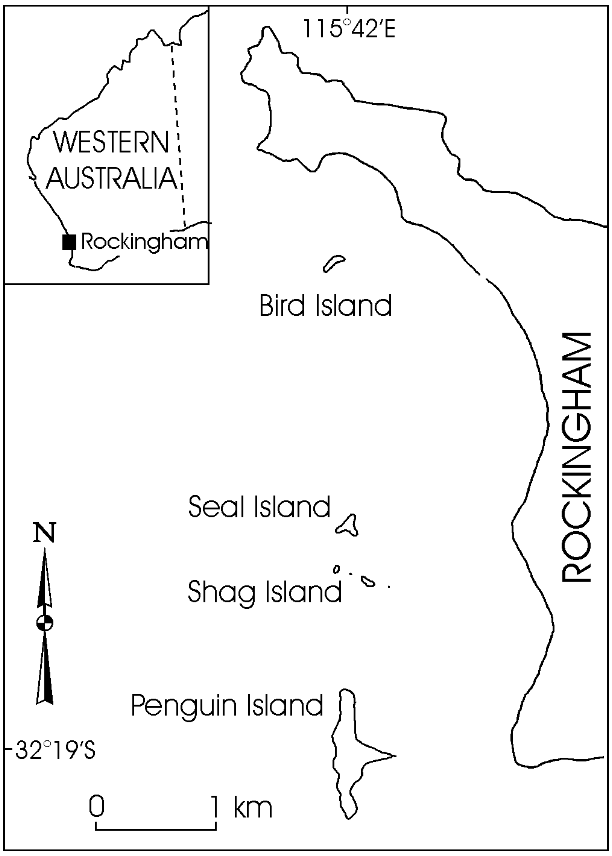

Fig. 1 The Shoalwater Islands off the coast of Western Australia

The problems

We were concerned about the changes in the vegetation pattern on the smaller islands for three reasons:

- There was a loss of biodiversity chiefly at the expense of native species. Notably, the Australian native holly-

Pages 254-259 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

hock (*Malva australiana*) appeared to be becoming locally extinct.

- Habitat change due to the thickets of European tree mallow (*M. dendromorpha*) could render the islands unsuitable for some nesting seabirds. Crested and Caspian terns that nest in the open are closed out by the canopy, and little penguins (*Eudyptula minor*) and bridled terns (*Sterna anaethetus*) are excluded by the lack of undergrowth.
- The risk of erosion was increased by the loss of native perennial shrubs which are constantly green and stabilise and maintain the shallow sandy topsoil. Tree mallow tends to die off leaving bare earth. Unstable soil is also unsuitable for burrowing birds.

The birds on the islands

Eleven species of seabirds have been reported nesting on these islands (Rippey *et al.* 1998). Little penguins, little shearwaters (*Puffinus assimilis*) and white-faced storm-petrels (*Pelagodroma marina*) nest in burrows or natural cavities among rocks or vegetation. A variety of terns nest on the surface of the ground; crested terns (*Sterna bergii*), bridled terns, Caspian terns (*S. caspia*) and occasionally roseate terns (*S. dougallii*). Silver gulls (*Larus novaehollandiae*) nest on open soil between plants. Pied cormorants (*Phalacrocorax varius*) nest, by preference, on top of perennial shrubs, especially *Nitraria billardierei.* Pelicans (*Pelecanus conspicillatus*) have nested on the northern promontory of Penguin Island since 1998. Feral pigeons (*Columba livia*) now nest in rock crevices on all of the islands.

Large amounts of guano are deposited by these birds. Pied cormorants and pelicans are the major guano-producing species of the region, and their numbers have tripled on these islands in the last ten years (E. Rippey, pers. obs.; Orr and Pobar 1992). Guano has raised the phosphorus levels of the soil of some of these islands to approximately 10%, the level of commercial fertilisers. The sands of the adjacent mainland shores are low in nutrients; Quindalup sands have phosphorus levels of 170 - 290 mg/kg (McArthur 1991), that is 0.017% - 0.029%.

Table 1 Reduction in total number of native and introduced plant species on Shoalwater Islands since 1975.

Island	Number of species 1975[1]	Number of species 1997[2]	Percentage Reduction
Penguin	81	76	6%
Seal	32	17	47%
Bird	31	14	55%
Middle Shag	22	16	27%

[1] (Abbott and Black 1980)
[2] (Rippey *et al.* 1998)

Fig. 2 Tree mallow (*Malva dendromorpha)* on Bird Island in October 1999 illustrating the lack of understorey.

The vegetation

There had been a marked diminution in the number of plant species found on the three smaller islands since the previous survey carried out in 1975 (Abbott and Black 1980; Rippey *et al*. 1998).

During the same period European tree mallow (*Malva dendromorpha*, formerly *Lavatera arborea*) (Ray 1995) had spread to all of these islands. The 1997 survey found it growing in profusion on the three smaller islands, forming dense stands of closely-packed upright stems up to 3 m tall (Fig. 2).

Four species of *Malva* are found on the islands, the Australian native hollyhock (*Malva australiana*) of the island variety (formerly *Lavatera plebeia* var. *tomentosa*) (Ray 1995), and three introduced species: European tree mallow (*M. dendromorpha*), marshmallow (*M. parviflora*), and Cretan mallow (*M. linnaei*). This last species will not be discussed in this paper as it currently occurs very infrequently on the Shoalwater Islands.

The native hollyhock and the tree mallow are similar and appear to hybridise (Rippey and Rowland 1995) although the hybrid seems to be sterile (E. Rippey, pers. obs.). Tree mallow, however, germinates earlier and has a less marked dormancy than the native hollyhock, which gives it an early advantage. This exotic species is also larger and grows more rapidly and profusely than the native hollyhock. It can completely overshadow any other vegetation and nothing will grow beneath it.

Tree mallow (*M. dendromorpha*) originated in the Mediterranean region and is found in coastal situations, often at the top of cliffs or on islands in association with bird colonies or in disturbed areas (Cook 1996). It is salt-tolerant and can excrete salt through glands on the leaves although it does not require salt for growth (A. J. C. Malloch, pers. comm.). It requires high levels of phosphorus and nitrogen and hence thrives in soil with a high guano content. It is not tolerant of frost. It has spread widely across the globe and now occurs in coastal sites with a Mediterranean or warm temperate climate in the United States, Chile, South Africa, and along the south-western and southern coast lines of Australia.

It is a palatable plant, sometimes used in Europe for animal fodder (Laghetti 1998), and is not found on the larger islands off Perth (Rottnest and Garden Islands) where there are grazing wallabies. On Penguin Island growth is sporadic around bird nesting areas but there is not massive overgrowth as there is on the smaller Shoalwater Bay islands. On some other small islands (Dyer Island and Green Island) in the Perth region, dense stands of tree mallow can be found, and it is also present on Carnac Island.

Tree mallow has fruits that drop to the ground beneath the plants (barochorous dispersal (E. Vidal pers. comm.)). They are too heavy to be blown over a great distance. Fruits can float in seawater for long periods and germination can take place following prolonged immersion in sea water (Ray 1995). The seed has a hard impermeable outer case (or testa) and can remain viable for many years. Silver gulls, numerous on all the affected islands, probably transport the seeds.

The native hollyhock (*M. australiana*) which grows on islands only occurs where colonies of birds have greatly enriched the soil with nitrogen and phosphorus (Yugovic 1998). It can be found on islands along the western and southern coasts from Dirk Hartog Island in Western Australia around to South Australia, and Victoria, and in the Bass Strait.

The third member of the Malvaceae family that is common on the smaller islands is the marshmallow (*M. parviflora*). This is a smaller annual herb growing up to 1 m in height. The plant is widely distributed in Western Australia along roadsides, and in stockyards as well as on the islands. It dies off and dries out in the summer. This is another introduction to Australia from the Mediterranean which grows on enriched soil (Hussey *et al.* 1997; Marchant *et al.* 1987).

Distribution of Malvaceae on the islands

Tree mallow is a relatively recent introduction. It was recorded on Bird Island in a survey carried out in 1959 but was not then present on the other islands (Storr 1961). By the time of the next survey 16 years later, it had reached all of the islands (Abbott and Black 1980).

The Australian native hollyhock of the island variety has almost disappeared from the islands. In 1998 the last remaining plant on Bird Island died and in 1999 the last one on Seal Island. There was no native hollyhock on Penguin Island, and less than 100 plants on Middle Shag Island.

In the spring of 1999 on Bird Island, tree mallow occupied the plateau of the island to the virtual exclusion of other species (save for a few alien grasses and herbs, and some *Nitraria billardierei* on talus slopes) occupying approximately two thirds of the plateau surface. Much of the eastern plateau is kept bare by pied cormorants that roost there.

On Seal Island a large stand of tree mallow occupied 20% of the plateau.

On Middle Shag Island, tree mallow covered the central area of the plateau, occupying about half of the area, and there was a somewhat smaller area of native hollyhock which was growing much more sparsely as scattered plants around it. Tree mallow grew at a density of up to 30 plants per m^2 (first-year plants) but just two plants per m^2 for the larger second-year plants.

METHODOLOGY

We aimed to reduce or eradicate the growth of tree mallow and restore native hollyhock and other native plants on Seal Island. This was the largest of the small islands and some clumps of native vegetation remained. It was used by more species of seabirds for nesting than the other small islands. There was also an area of beach, facilitating access by boat.

Working parties of volunteers from the Friends of the Shoalwater Islands Marine Park visited Seal Island under the supervision of a ranger from the Department of Conservation and Land Management on the following dates:

11 September 1998: 10,000 tree mallows removed. Larger plants cut down and 15% glyphosate applied to stumps. Smaller ones uprooted by hand.

13 June 1999: 8000 tree mallows removed. Weed mat (5 m x 5.4 m) laid and planted with young native hollyhock plants.

3 July 1999: Native seedlings planted in and around cleared area: (*Rhagodia baccata, Myoporum insulare, Carpobrotus virescens*).

5 March 2000: 1200 tree mallows removed.

14 May 2000: Weed mat laid down and planted with *R. baccata.* Native hollyhock (six plants) planted in small (1 m x 1 m) weed mat squares. 1 m x 1 m areas adjacent to small weed mat squares tilled by turning over with a spade.

13 August 2000: 1800 tree mallows removed. Planted native hollyhock (three plants), *Frankenia pauciflora*, and *Rhagodia baccata* in small numbers. Additional unsupervised visits were made to measure growth rates and to assess the success of plantings.

RESULTS

Cutting and stump poisoning of tree mallow reduced the population from 10,000 to 3000 i.e. (70% reduction). Uprooting plants was less successful as plants could re-root to produce flowers and fruit. New germination follows soon after cutting and uprooting and growth is rapid at more than 1 cm a day during the spring months. We tried to cut down new plants before they could fruit. Although flowering and fruiting usually occurs in the second year, a variable percentage (at least 5%) flower in the first year. Flowering takes place from August to October and fruit sets in November through to January, so removal is best carried out in July or August. However, new growth can occur from seed banks in the soil in later spring and a further session of removal in summer may be required. The seed bank is vast and *Malva* seeds are long lived, capable of germination after a century (J. Conran pers. comm.; Spira and Wagner 1983). Reducing the number of tree mallows was not enough to allow the native vegetation to re-grow; other weeds, such as the annuals *Lolium rigidum, Hordeum leporinum, Urtica urens, Chenopodium murale, C. album*, as well as the smaller marshmallow (*M. parviflora*) came in to take its place.

The weed mat was laid to prevent growth of tree mallow and other weeds in one area and to allow planted native species to grow without competition. This worked well for native hollyhock which grew in the mat from the first planting and then re-grew apparently from seed shed on to the degrading weed mat.

Tilling was undertaken because marshmallow (which is an agricultural weed of no-till farming) is controlled by cultivation (Anon 1999). Perhaps this is explained by the fact that Malvaceae seeds need to be on or very close to the surface for germination to take place (Okusanya 1979). The small experimental tilled areas on Seal Island remained clear of growth for some three months, well into spring.

Planting of natives (such as *Rhagodia baccata, Frankenia pauciflora* and *Myoporum insulare*) was not successful with no long-term survivors whether planted randomly or in weed mat. However *Carpobrotus virescens* planted on a rocky cliff top thrived.

In 2000, crested and Caspian terns nested on the island in considerable numbers, in areas formerly occupied by *M. dendromorpha.*

DISCUSSION

Assessment of the current situation on Seal Island

It seemed as though our efforts were not entirely in vain. Tree mallow was being reduced and there had been some regrowth of native hollyhock. However, revegetation with native species did not appear to be taking place and re-planting with seedlings and young plants was notably unsuccessful. Weed mat was useful in preventing the excessive growth of weeds locally but only native hollyhock had been grown successfully in it, and it was an expensive way of controlling weeds.

Tilling or shallow digging over the surface seemed to work almost as well in clearing and preventing the rapid regrowth of weeds and was easier, quicker, and cheaper to implement over a wider area. Surface-nesting birds were returning to breed on the island.

Plans for future management would include:

- Continuing removal of tree mallow by hand, at least twice a year.
- Surface tilling in selected areas to reduce growth of annual weeds followed by replanting in tilled areas
- Collecting seed of native plants and hand seeding in both tilled and untilled areas.

The status of tree mallow in Australia and overseas

Tree mallow does not appear to present the same problem elsewhere as it does on the Shoalwater Islands. In the Mediterranean and in Western Europe it is appreciated for its appearance, and as it is palatable may even be used for animal fodder (Laghetti 1998).

On islands off Marseille it can grow densely in areas and a local researcher is worried because it represents an important source of water for rats (*Rattus rattus)*, which then pose a threat to the shearwater population (E. Vidal pers. comm).

British bird conservationists have used *M. dendromorpha* to provide shelter for nesting roseate terns (*Sterna dougallii*) (Avery *et al.* 1995)

In the Farallon Islands off the coast of California it is appreciated because it provides habitat for migrating land birds, but growth is controlled (C. A. Morris pers. comm.).

In South America and South Africa it grows sporadically along the west coasts, but is not a cause of concern.

State herbaria and parks and wildlife authorities in Australia were contacted by telephone. There were two areas where tree mallow had become so dominant that authorities had felt compelled to take action to control it: Mud Island in Port Philip Bay, Victoria and West Island off the coast at Victor Harbour in South Australia.

Mud Island with a land area of about 51 ha consists of Quaternary shifting sands around a lagoon. West Island, 12 ha, consists of ancient granites. Given that Seal Island, 1.2 ha, is formed of limestone, it appears that underlying substrate is not a significant influence on the growth of tree mallow.

All of the islands are used intensively by nesting birds. On Mud Island there are some 5, 000 silver gulls and about 15,000 pairs of ibis. Straw-necked ibis (*Threskiornis spinicollis)* and Australian white ibis (*T. molucca*) have nested there for the past 10 years (Yugovic 1998). On West Island large breeding colonies of little penguins, silver gulls and crested, fairy (*Sterna nereis*), and Caspian terns are found (Robinson *et al.* 1996).

All the islands have frost-free climates with rainfall concentrated in the winter, although the amounts vary considerably (Mud Island - 612 mm; West Island - 450 mm; Seal Island - 800 mm).

These islands have a history of disturbance. Mud and West Islands were quarried for guano and granite respectively. Huts have been built on them. Rabbits had grazed on both for a century or more and were eliminated only 20 or 30 years ago.

The overwhelming growth of *M. dendromorpha* appeared to be triggered on Mud and Seal Islands by the arrival of unprecedented numbers of nesting birds: ibis on Mud Island, pied cormorants on Seal Island. These are large birds that deposit a great deal of guano.

Eradication efforts on Mud Island

By 1994 over half the land area (30 ha) was occupied by tree mallow. Action was indicated to restore the habitat for nesting birds (particularly for crested, Caspian and fairy terns), and also to protect the Australian native hollyhock. In 1994 one ranger with Parks Victoria and a group of eight to twelve volunteers started monthly visits to Mud Island. They cut down the tree mallow, at first in thousands, using chainsaws to sever the 15 cm thick woody trunks. They treated the large stumps with glyphosate, and scattered the seeds of native hollyhock. Now after seven years they find only a few tree mallow seedlings on each visit and the island has large meadows of native hollyhock (Yugovic pers. comm.).

Eradication efforts on West Island

In South Australia the Department of Natural Environment and Resources initially undertook a more scientific approach. In 1994 and 1995 they trialed 10 plots with burning, cutting and spraying with the herbicide Brush-off (metsulfuron).

All approaches over the two years were effective in controlling the growth of young plants of *M. dendromorpha*, but it was apparent from continuing new germinations that the seeds were long-lived in the soil and that a long-term approach would be necessary. Some treatments although killing current growth, seemed to stimulate subsequent germination. Researchers also scattered seeds of native grasses and early observations showed successful germination. With courageous lateral thinking the introduction of tammar wallabies (*Macropus eugenii*) was considered (as mentioned the tree mallow is palatable), but black-footed rock-wallabies (*Petrogale lateralis*) introduced earlier had died out for lack of water on the island. Other biological control measures suggested were rabbits, goats or the native greater stick-nest rat (*Leporillus conditor*).

Scarcity of funds and labour have led to the abandonment of efforts to rehabilitate West Island.

Problems in management of invasive tree mallow

Lack of information

Perhaps the three separate groups attempting to control *M. dendromorpha* over seven years would have benefited from sharing information with each other. We also spent time determining the status of tree mallow in Australia and around the world.

Information about control measures is as important as knowledge of areas at risk. It seems that all seabird nesting islands on the southern coasts of Australia could be at risk. The growth of native hollyhock on these islands may be a marker for those at greater risk. Early identification of the problem could allow eradication of tree mallow at relatively little expense before it has become established.

Access

Islands mean difficult and expensive access; either a boat or a helicopter is required and some sort of landing area such as a sheltered beach is needed if there is to be all-weather access. It can add to the costs considerably if a boat has to be hired (for example A$120 to hire a ferry to transport weed mat to Seal Island; A$300-400 to hire a boat for each visit to West Island). Many domed granite islands off the south coast are accessible only by helicopter.

The presence of breeding birds can also restrict access as they can be very sensitive to disturbance. In the Shoalwater Bay Islands nesting terns can prevent access for 2-3 months in the spring and cormorants for a similar period in the autumn. Little penguins occupy their burrows for many months, from April to December in Western Australia (Pizzey 1997).

Funding

Programmes may be difficult to fund where offshore islands are seldom visited by ratepayers or voters, especially if landing is prohibited. Seabird nesting islands may not be regarded as having a high priority.

Labour

Weed control is labour intensive and long term but can be successful and rewarding as shown by the efforts on Mud Island. Here a dedicated ranger and a group of enthusiastic volunteers together with a researcher with vision, were responsible for the sustained effort which has resulted in the restoration of the native vegetation. On West Island by contrast, funds were not available, costs were high, volunteers were not considered, and the project was dropped. Using volunteers is probably the only way such programmes can be afforded. Where volunteers are used it is important that the controlling authority should be supportive both in devising a workable plan with the best available advice, and in the long-term execution of the task.

CONCLUSION

The weed tree mallow (*M. dendromorpha*) can be successfully contained simply by mechanical removal at regular intervals over a long period. Subsequent re-colonisation by Australian native hollyhock (*M. australiana*) can be facilitated by planting or seeding.

Eradication of tree mallow is not a realistic aim on the Shoalwater Islands. There is a large bank of long-lived seeds in the soil. In addition, seed rain comes from scattered plants along the metropolitan coast line, from plants growing on inaccessible cliffs on the islands, and from other islands in the vicinity that are more costly and risky to access. Some of these islands are larger and further from the mainland, some are surrounded by cliffs that have to be climbed before any plants can be reached, and some are infested with tiger snakes (*Notechis scutatus*). Thus constant vigilance will have to maintained on the Shoalwater Islands.

The wide range of annual weeds that have arrived on the islands, including marshmallow (*M. parviflora*) seem to pose less of a problem. Perhaps tilling and seeding with natives could help in the re-establishment of the original vegetation in areas where this has disappeared. However, true rehabilitation is unlikely as long as the number of birds, especially cormorants, nesting on the islands remains at present levels.

The value of vigilant volunteers has been illustrated by their role in both the early discovery and control of tree mallow.

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Practical concerns in the eradication of island snakes

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Abstract Successful eradication of the introduced and invasive brown treesnake (*Boiga irregularis*) from two 1 ha areas on Guam led us to suggest that the snakes could be eradicated from large nature reserves if immigration of snakes from adjoining areas could be eliminated or greatly reduced with perimeter snake barriers. Practical problems encountered in the design of snake barriers on Guam include the extraordinary climbing abilities of brown treesnakes, high levels of rat damage to chewable barrier surfaces in snake-reduced areas, and frequent and destructive cyclonic storms. Four successful snake barrier designs have been developed, and one 23 ha site on Guam has been largely trapped out following erection of a snake fence around the perimeter. Unresolved problems include the failure to capture all snakes within the 23 ha exclosure, and the fragility and high maintenance requirements of low-cost barriers. Our attempt to use brown treesnake traps for control of introduced wolf snakes (*Lycodon aulicus*) on Ile aux Aigrettes, Mauritius was unsuccessful, possibly due to low snake densities, size selectivity of the traps, or seasonal cessation of feeding activity.

Keywords snake eradication; *Boiga irregularis*; Guam Island; snake exclosure; *Lycodon aulicus*; Ile aux Aigrettes; Mauritius; trap selectivity.

INTRODUCTION

Our experience with eradication of island snakes derives primarily from study of the brown treesnake (*Boiga irregularis*) on Guam. Aside from a tiny, subterranean termite-eating worm snake, the remote oceanic island of Guam had no snakes prior to arrival of the brown treesnake shortly after World War II (Savidge 1987; Rodda *et al*. 1992a). In the half century following arrival of the snake, Guam lost virtually all of its native forest vertebrates, including 10 of 13 birds (Savidge 1987), two of three mammals (all bats), and half of its 10-12 lizards (Fritts and Rodda 1998). In addition, some wetland birds disappeared or declined inexplicably, sea birds ceased nesting on Guam, and a large number of the introduced forest animals declined in abundance. The causes of extinction are rarely clear, but the commonality in most of these declines was the unprecedented level of predation each species experienced due to the snake. We judge that most of the bird declines and perhaps half of the lizard losses are attributable to the snake (Rodda *et al.*1997, 1999c). The circumstances surrounding the loss of the bats are shrouded in mystery (Wiles 1987). The snake undoubtedly played a role, but human persecution may also have been a contributing factor (Wiles *et al*. 1995).

One commonality among these extinction stories is that the prey species lacked co-evolutionary experience with snakes (Rodda *et al.* 1999c). An anecdote will illustrate this familiar point. During the course of her avian disease studies, Julie Savidge (Savidge 1987) maintained an aviary with bridled white-eyes (*Zosterops c. conspicillatus*), a diminutive flocking bird that roosts communally. A brown treesnake gained entry to the aviary one night and was discovered while preying on the birds, which were perched immediately next to each other in a row. Lacking co-evolutionary experience with a nocturnal arboreal predator, the unconsumed birds remained in place on the branch as their neighbours were eaten (Jaffe 1994).

This phenomenon, sometimes called island tameness, is characteristic of islands lacking mammalian predators. Thus insularity was a contributing cause to the ecological catastrophe that happened on Guam when the snake arrived. On the other hand, insularity also made it practical to keep the problem from spreading. The U.S government, though its Wildlife Services agency, has embarked on a rigorous programme to keep the snake from spreading to other places (Oldenburg and Worthen 1997). Had Guam been part of a much larger landmass, the snake's spread would have been difficult or impossible to contain. For example, in 1993 a brown treesnake reached Corpus Christi, Texas (McCoid *et al.* 1994). Had the brown treesnake become established in coastal south Texas, what would have blocked its spread from there throughout the southeastern U.S and possibly the Neotropics?

CREATING INSULARITY

Guam's wildlife suffered catastrophic loss when their protective insularity was breached by human introduction of an alien predator. However, humans can also restore insularity by creating artificial islands of snake-free habitat. Specifically, we have found it possible to create small, predator-free nature reserves using a combination of snake barrier and eradication methodologies (Rodda *et al.* 1999a). The first example of this was Campbell (1996), who eliminated brown treesnakes from two 1 ha snake

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exclosures and compared the densities of prey species in the year following snake removal to those of similar but snake-occupied 1 ha plots nearby. There were no birds or bats present in his study site, so changes in those populations could not be detected. The remaining lizard species, however, showed a dramatic response. Within a year their numbers roughly doubled (Campbell 1996). It would be easy to understate the magnitude of the accomplishment of building an effective snake barrier. Most snakes are good climbers; the brown treesnake is one of the very best.

Campbell's work showed that snake removal and wildlife restoration were possible, but it did not show that they were practical. To be practical the cost has to be within reason, the protected area has to be large enough to support viable populations of the prey species, and the barrier must be durable enough to withstand challenges by humans and natural forces. The Campbell barriers brought attention to two acute problems: typhoons and rats. Rats chew holes in all things chewable, particularly barriers that bisect their home ranges. A larger problem is that Guam is subjected to irregular but severe cyclonic storms. For example, in December 1997 Supertyphoon Paka pummelled Guam with steady winds of up to 265 kph, and with gusts topping out at around 380 kph (from news reports). During the 1990s, Guam was subjected to 15 typhoons, of which about half had sustained winds over 150 kph (based on our list compiled from reports of the US Naval Oceanographic Command/Joint Typhoon Warning Center). Thus to protect wildlife from brown treesnakes in perpetuity on Guam, a snake barrier must be extremely durable.

Over the past decade we have studied barrier effectiveness and durability (Perry *et al*. 1996, 1997, 1998; Rodda *et al.* 1998; Campbell 1999). Barrier designs are tested progressively through three types of challenges. First, we build a door-sized mock-up of the design in the wall of a laboratory test chamber. Snakes attempting to escape from the test chamber are videotaped under infrared illumination in total visible-light darkness to determine the mechanism of escape, if any. Barriers that pass this test progress to the next stage, in which we confine snakes in a small octagonal enclosure built entirely of the proposed design. If the number of escapes is trivial or zero, the design is then tested in a large outdoor enclosure that we stock with a high density of snakes (for methodological details see Perry *et al.* 1996, 1997, 1998; Rodda *et al*. 1998; Campbell 1999). In brief, we have identified four classes of successful designs: temporary, bulge, masonry, and vinyl. Temporary barriers are used for interdicting snakes in commerce; they are not suitable for restoration of endangered species. Bulge barriers are retrofitted on a chain-link fence, and are therefore vulnerable to damage by strong typhoons, though they have been used as a low-initial-cost alternative to more permanent designs. The vinyl barrier uses material designed for long-term use as seawall; it is mechanically durable, but we have some unresolved concerns that the surface finish may degrade over time in the Guam environment. Surface finish must remain smooth to keep snakes from climbing the barrier. Our favoured masonry material is a pre-stressed moulded concrete design that is 100% successful in repelling snakes, and impervious to rat and typhoon damage, but has a fairly high initial cost (*c.* USD300/m). A conservative life expectancy of fifty years for the concrete barrier makes the cost reasonable (USD6/m/y), but it is challenging to pay for this entire cost "up front."

FIRST WILDLIFE APPLICATION ON A SNAKE EXCLOSURE

One practical experience in the use of such barriers is the 23 ha patch of forest on Andersen Air Force Base that is surrounded by a bulge barrier exclosure and is generally known by its military designation, "Area 50." The Area 50 project has been managed by Guam's Division of Aquatic and Wildlife Resources (GDAWR), with technical assistance and funding provided by a variety of federal agencies (US Geological Survey, US Fish and Wildlife Service, Wildlife Services). Snake control in Area 50 was initiated prior to construction of the barrier in 1997 (Searle and Anderson 1998). Sixteen radio-collared Guam rails (*Gallirallus owstoni*) were released in the area in 1998, when the snake population had been reduced but not eliminated (Beauprez and Brock 1999). Snakes continue to be caught in the area; persistent capture rates vary from zero to seven snakes per week (Diane Vice, GDAWR, pers. comm. 2001). Guam rails are federally listed as endangered. Except for Area 50, they are extinct in their native range (endemic to Guam), though a small extralimital population has been established on the nearby snake-free island of Rota. Because they are essentially flightless, they are exceptionally vulnerable to terrestrial predators, though they are agile and fecund, and adults have some ability to defend themselves against brown treesnakes, at least during the day. The fate of the 16 rails in Area 50 has not been established, but some survived (an average of 198 days, with five birds alive at the end of the 318 day reporting period: Beauprez and Brock 1999), and some have been recovered from feral cat stomachs (R. Beck, GDAWR, pers. comm. 2000). One problem with a fenced artificial island such as Area 50 is that the fence can be used by a clever carnivore such as a cat for assistance in capturing flightless birds. In the future we will conduct multi-species predator tests of barriers.

More troubling to us is the persistence of snakes in Area 50. After four years of nearly continuous trapping, substantial numbers of snakes are still being captured in Area 50. Our tests on smaller exclosures (Campbell 1996) indicated that snakes could be eradicated, not merely depressed in abundance, from snake exclosures. Is there some attribute of snake capture that does not scale up in going from 1 to 23 ha exclosures? Or is the barrier used in Area 50 allowing penetration by snakes? Unfortunately, there is no obvious way to identify the source of snakes that have been captured inside Area 50. Nine percent (seven) of 78 marked snakes released outside the area after the barrier was completed were subsequently captured inside

(Searle and Anderson 1998). Thus, some penetration has occurred. But are all or the majority of snakes invaders? Opinions differ, and direct evidence is lacking because the nature of the conservation activity in Area 50 precludes release of marked snakes.

Snakes encountered inside Area 50 could have: (1) breached the barrier, (2) grown up inside the exclosure, or (3) been present as adults inside the exclosure throughout the trapping period (i.e., refractory to trap capture). The reason for distinguishing the latter two conditions is that traps are known to have difficulty capturing small snakes (Rodda and Fritts 1992; Rodda *et al.* 1999b). It is less troubling if the failure to capture is due to a known phenomenon such as reduced success in trapping small snakes. Failure to capture an adult snake is a new phenomenon.

Barrier breaches can occur because the design of the barrier is deficient, the construction is defective, or the maintenance is inadequate. During our laboratory tests of the bulge barrier we tested both a meticulously constructed version and a degraded one (Perry *et al.* 1996, 1998). To create the degraded version we added an additional flat piece of hardware cloth to the base of the barrier. The exposed tines of the cut mesh provide numerous minute edges that a snake can use to climb partially up the barrier. The added hardware cloth layer simulated the edges that are present in the seams of bulge barriers that are poorly made. The carefully constructed version stopped 99% of 344 escape attempts, and all of the escapees were unusually large individuals (total length >2200 mm) that could reach over the bulge from the ground (Perry *et al.* 1998). Snakes of such a size constitute less than 1% of the population (Fritts 1988; Savidge 1991; Rodda *et al.* 1999d) and are all male, so they could not re-establish a snake population by themselves. On the other hand, degraded bulge barriers are relatively easily climbed by even small snakes. Given enough time, only 26% of the ordinary sized snakes (total length <1500 mm) failed to escape from an enclosure built with a degraded bulge barrier. This is one of the reasons why we do not recommend the use of this design for nature preserves (Perry *et al.* 1998). The bulge barrier design is not robust; it does poorly if construction is substandard. However, it is attractive to programme administrators because it has a low initial cost.

Another source of difficulty may be the gate that is included in the Area 50 barrier. Gates in exclosures are always problematic and they are difficult to test realistically in a small controlled environment. In our laboratory and field tests of exclosures we omit gates, as realistic gate results depend on site-specific details. The main defence against gate breaches is to deflect travelling snakes away from the gate. The gate used in Area 50 is located in an ideal place (maximally removed from any adjacent trees), but it does not have deflectors, and any snake approaching it would pass through easily.

The calibre of construction on the Area 50 barrier did not conform to our laboratory standards, so the effective breach rate is probably somewhat intermediate between results of the laboratory tests for the meticulously constructed version and the degraded one. Maintenance has also been irregular, facilitating breaches primarily through the growth of vegetation on or through the fence. In addition, oxidised fence components have not always been replaced in a timely fashion, and on occasion animal parts, such as preying mantis egg cases, have been allowed to remain on the fence, providing a purchase for climbing snakes. The frequency of these problems underscores the lack of robustness in the barrier design. To borrow a sporting metaphor, there is no depth to the defence. Unless a great deal of effort goes into quality control and maintenance, snake repulsion will be compromised.

SELECTIVITY IN TRAP CAPTURE

If all snakes are vulnerable to capture, those that breach the fence should eventually be caught, as some level of snake trapping has occurred in Area 50 since its construction in 1997. At that time it was known that small snakes were unlikely to be caught. This conclusion was based on the relative failure of traps to capture snakes smaller than about 800 mm SVL (snout-vent length; Rodda *et al.* 1992b, 1999b). Brown treesnakes hatch at around 300 mm SVL (Fritts 1988; Rodda *et al*. 1999c). Hatchling brown treesnakes are relatively easily sighted, however, so the Campbell (1996) project relied largely on visual searches to ensure that snakes of all sizes had been eliminated from the 1 ha exclosures. Visual searches are relatively tedious and time consuming, however, and were not used for eliminating snakes from Area 50. Instead the managers of that project chose to rely on growth of hatchling snakes to trappable size.

One issue potentially affecting capture probability is the long-term effect of lethal control of snakes using snake traps. Wildlife Services maintains 2000-3000 snake traps on Guam, primarily as a deterrent to snakes spreading to other islands. All snakes captured are killed. While this is highly desirable, it runs the risk of selecting for snakes that are refractory to entering traps. If there is genetic variation in propensity to enter traps, this continuous lethal control may be inducing selection for trap avoidance. In the vicinity of Area 50, however, lethal control has been relatively short term, so it is not yet likely to be a concern.

Another concern is the potential for prey abundances to sharply increase in any effective snake exclosure. As illustrated by Campbell's (1996) study, prey may become more numerous in areas depleted of snakes. Any hungry snake present inside a snake exclosure would then have the option of dining on either the abundant prey present in the area, or entering a trap to get close to the food attractant in the trap (all successful brown treesnake traps to date have relied on a food attractant: Rodda *et al.* 1999b). Thus high prey abundances may depress snake trapping success. This problem may also affect efforts to eradicate an incipient population on a prey-rich island such as Saipan, where numerous brown treesnake sightings have been reported (Fritts *et al.* 1999).

How successful are brown treesnake traps? In relation to literature values on capture success, they are the most successful snake traps known (see Fig. 20.5 in Rodda *et al.* 1999b). The literature values are based on captures per trap night. For the purposes of eradication, however, the key statistic is captures per snake present. Based on our traps for brown treesnakes on Guam, we have captured between about 1% and 25% of the snakes present per night (as determined by open population mark-recapture models: Rodda *et al*. 1999d), with a long-term average of about 12.5% (G. Rodda and K. Dean-Bradley unpub. data). Such a high rate of capture, if it applies to all individuals, should permit the elimination of a population in a few weeks (Rodda *et al*. 1999a).

The 12.5% figure is an average, of course. Is it possible that some snakes are more easily captured and some snakes less so? On logical grounds one would assume so; there is presumably some inter-individual variation in capture vulnerability. More troubling, are there some snakes that are totally refractory to capture? Inter-individual variation in capture vulnerability is relatively easy to quantify if one has a closed population (no ingress/egress/births/deaths). In such a case one can assume that all animals detected at any time were present throughout, allowing precise estimation of their individual capture probability. If the snakes are free to come and go, however, one cannot rigorously distinguish capture probability from the probability of their being in the area. We have found no areas in Guam that are of a practical size for mark-recapture trials and that are demographically closed. This has stymied efforts to quantify individual heterogeneity in capture probability.

Using a variety of trapping studies of our own (Rodda *et al*. 1992b, 1999a,b), we were able to quantify the capture probability of size classes of snakes. We pooled 21 trap history matrices into one large matrix involving 942 individual snakes divided into five size groups by snout-vent length (SVL) (601-700 mm, 701-800 mm, 801-900 mm, 901-1000 mm, and >1000). This pooling created a matrix of limited value for estimating survivorship or other population values, but it maximised our ability to discern capture probability differences among size classes. We used the program MARK's (White and Burnham 1999) open population model (Cormack-Jolly-Seber) to evaluate models involving group and time effects on both "survivorship" (φ, effectively 1 - emigration rate between daily capture occasions) and capture probability (*p*). This analysis revealed no relationship between snake size and survivorship, but it did indicate a strong relationship between size and capture probability (Fig. 1). No snakes below about 600 mm were captured, supporting earlier observations (Rodda *et al.* 1999b).

In the size range 600-900 mm SVL capture probability increases sharply, to a maximal value for snakes 900-1000 mm SVL (brown treesnakes mature in this size range: Rodda *et al.* 1999c). We are testing new trap designs to capture small snakes. In the meantime it should still be possible to eradicate a closed population of brown treesnakes if the smaller snakes are captured as soon as they reach a trappable size. It is not known how long this will take. In captivity, well-fed snakes reach a capturable size in about one year, but growth rates of juveniles in the wild are unknown.

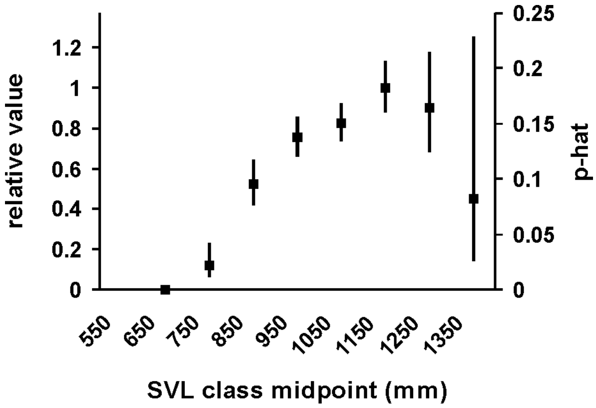

Fig. 1 Capture probability for brown treesnakes on Guam using conventional mesh-flap traps. Analysis is by open model (Cormack-Jolly-Seber), with snake body size defined by snout-vent length (SVL) in 100 mm size classes.

One puzzling result of brown treesnake reproductive studies (F. J. Qualls and C. P. Qualls, unpub. data; Aldridge 1996, 1998) is that reproductively active males appear to be relatively rare. This is surprising, because female reproductive activity occurs at all times of year in brown treesnakes (F. J. Qualls and C. P. Qualls, unpub. data; Rodda *et al*. 1999c). From an adaptive perspective, one would expect males to be able to take advantage of mating opportunities at whatever time of year they encounter a receptive female. Yet reproductively-active males are relatively rare in samples of brown treesnakes (which are collected primarily with food-baited traps). One possible explanation for this phenomenon might be that snakes that are reproductively active are refractory to trap capture. Snake breeders report that male snakes in general avoid eating while they are in reproductive condition (N. Ford, pers. comm.). Females are also refractory to feeding while gravid. Neurochemical studies of the brains of reproductive red-sided gartersnakes (Morris and Crews 1990) indicate that a specific brain chemical (neuropeptide Y) acts both as a feeding inhibitor and reproductive inducer in that species. Thus, reproductive aphagia might account for some of the variability we have seen in capture success, and it might indicate that some individuals are totally refractory to trap capture at certain times. It is not known what role, if any, neuropeptide Y has in brown treesnakes.

ILE AUX AIGRETTES, MAURITIUS

Despite the difficulties we have identified in eradicating snakes from Area 50, we were able to eradicate snakes from the 1 ha (Campbell 1996) study sites. Average trap capture probabilities of 10-20% per night suggested that if barrier leakage was not a problem, eradication should be completed in a few weeks. We were offered an opportunity to test this concept on the island of Ile aux Aigrettes off the east coast of Mauritius, Indian Ocean. The wolf snake (*Lycodon aulicus*) was introduced to Mauritius around 1860 (Cheke 1987), and it no doubt spread to the offshore islet of Ile aux Aigrettes sometime after that. It has been associated with the loss of several native lizards, so the Mauritius Wildlife Foundation elected to restore the islet by removing the introduced snake (C. Jones and S. Harris pers. comm.). We volunteered our trap design and tested it during a short visit to the island in December 1999. The 24 traps that we tested were alternately baited with day geckos (*Phelsuma ornata*), night geckos (*Hemidactylus frenatus*), or laboratory mice (*Mus musculus*). The traps were monitored for a period of about six weeks, during which they failed to capture a wolf snake. We saw one wolf snake during a visual survey and one was eventually found dead in a trap after trap monitoring was discontinued and the attractants were removed (Harris 2000).

Why did we fail to capture wolf snakes in our traps? Unlike our Guam trap experiments, for our work on Ile aux Aigrettes we were able to prepare only a small number of traps, and we have no information on the density of wolf snakes on Ile aux Aigrettes. Wolf snakes might be exceedingly rare, limiting the opportunities for even a single capture with so small a number of traps. The size selectivity of our traps (Fig. 1) might have worked against us, as the average size of a wolf snake is likely to be around 700 mm SVL (no wolf snake size data for Ile aux Aigrettes are available). Note that the size selectivity illustrated for brown treesnakes in Fig. 1 is for a flap trap baited with a mouse attractant; no comparable data exist for the open-cone trap type and attractants used for wolf snakes in Mauritius. Another possibility is that the time of our trapping on Ile aux Aigrettes happened to coincide with the wolf snake's mating season there, in which case capture success might be depressed.

The above-listed concerns appear to be the best candidates for understanding the incomplete success we have seen in elimination of snakes from Area 50 on Guam and Ile aux Aigrettes in Mauritius, but this list of possibilities is not exhaustive. We do not yet know whether the essential condition for eradication – removing snakes faster than recruitment is replacing them – can be met. It may be practical to eradicate invasive snakes from these nature reserves without rectifying these problems, but to accomplish eradication without solving these problems will undoubtedly increase the cost over that originally anticipated. Additional quantitative information on the severity of the problems and the costs of rectifying them will be needed to identify the optimal snake eradication strategy.

ACKNOWLEDGMENTS

As this paper summarises our experiences over more than a decade of studies, it would be impossible to name all of the individuals who have contributed to our efforts. Those acknowledged in the cited papers all also made an appreciated contribution to this paper. Primary funding has been supplied by the US Department of Defence Legacy Program and the US Department of the Interior's Office of Insular Affairs. The Guam Division of Aquatic and Wildlife Resources has not only hosted our efforts on Guam but also provided much of the data on Area 50. The US Department of Agriculture's Wildlife Services has contributed much insight into the effectiveness of snake trapping strategies. C. Jones and S. Harris (Mauritius Wildlife Foundation) were instrumental in engaging us for the preliminary effort at snake eradication on Ile aux Aigrettes. Diane Vice and Julie Savidge suggested improvements to the manuscript.

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An ecological basis for control of the mongoose *Herpestes javanicus* in Mauritius: is eradication possible?

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Abstract The mongoose (*Herpestes javanicus*) was introduced to Mauritius in 1902 to control rats and now threatens the native fauna. In the 1980s a Non-Governmental Organisation, the Mauritian Wildlife Foundation, began controlling mongooses in ecologically sensitive areas using labour-intensive grids of box-traps. As this is difficult to sustain in the long term, the ecology of *H. javanicus* on Mauritius was studied from 1997-2000 to suggest improvements to control methods and alternative techniques that could replace or augment current control methods. Using census techniques, radio-telemetry and long-term trapping, we found that mongooses are not territorial and achieved densities up to 50 animals/km^2 in some habitats. Home ranges varied from 0.25-1.10 km^2, with no significant seasonal or sexual variation. Degraded forest, riparian and rocky habitats are the most favoured habitat types. Although mongooses consumed birds infrequently, rare predation events have a significant impact on the numbers of the endemic pink pigeon (*Columba mayeri*). We discuss how the information from the study can be used to improve the management of mongooses using current methods of trapping, how alternative control methods can be adopted to enhance control, and whether eradication is achievable.

Keywords Small Indian mongoose, (*Herpestes javanicus*); Mauritius; pink pigeon; endemic birds; trapping; home range; habitat use; diet; census.

INTRODUCTION

The introduction of animals to island ecosystems often has deleterious consequences on indigenous fauna and flora (Atkinson 1996). This is particularly true if the species introduced is a carnivore with generalist feeding habits to which the native fauna is not adapted. The lesser Indian mongoose (*Herpestes javanicus*) is such a carnivore and has been introduced to many tropical oceanic islands, 70% of which fall within recently-designated biodiversity hotspot areas (Myers *et al*. 2000).

Mongooses were originally introduced to oceanic islands to control rats in sugar cane fields, but the species' ability to control rat populations is dubious as rats continue to thrive in areas where mongooses occur and can withstand a high degree of predation (Pimental 1955; Seaman and Randall 1962; Gorman 1975). At the same time, it is implicated in the decline of rare and endemic species from a wide range of taxa (Baker and Russell 1979; Honegger 1980; Nellis and Small 1983; Nellis *et al.* 1984; Coblentz and Coblentz 1985; Jones 1988). On Mauritius, mongooses are blamed for the local extinction of Audubon's shearwater (*Puffinus l'herminieri*) (Cheke 1987), introduced game birds (Cheke 1987), and ground-based skinks (Vinson and Vinson 1969; Jones 1988).

Conservation on Mauritius is of high priority. The island falls within one of the designated biodiversity hotspot areas due to its high levels of endemism (Myers *et al.* 2000). Through human agency it has lost more than half of its vertebrate fauna and 90% of its original vegetation cover (Cheke 1987), and has gained at least thirty alien vertebrate species. In collaboration with the Government of Mauritius, The Mauritian Wildlife Foundation (MWF) has been striving to conserve some of the rarer endemic species that still persist on Mauritius since the late 1970s (Jones and Hartley 1995). The management of invasive species is an important aspect of current conservation efforts on Mauritius, and the control of introduced predators, like the mongoose, is an important part of this programme.

Of the surviving large land birds of Mauritius, the pink pigeon requires the most management. The population size of this species was estimated at fewer than 20 wild birds

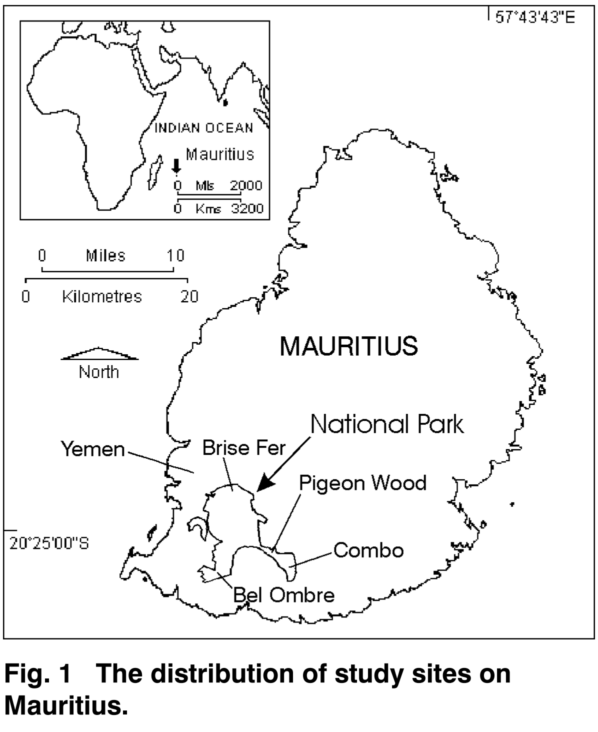

Fig. 1 The distribution of study sites on Mauritius.

Pages 266-273 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

in the late 1970s (Jones 1987). Due to captive breeding and re-introduction programmes begun in the late 1980s (Jones and Swinnerton 1997), the wild population now stands at over 400 birds. Birds have been released to four mainland sites within the National Park in south-west Mauritius (Fig. 1), and to one predator free offshore island not shown on the map. These sites were chosen by MWF because they were remnant areas of high quality native forest within the National Park, which could be easily managed. The pink pigeon naturally spends a lot of time on the ground (Roy 1994) and is vulnerable to introduced ground-based predators. Long-term predator control is a crucial component of the conservation of this species.

Mongooses are currently controlled in Mauritius by using simple box traps (Fig. 2) laid out in grid systems (Fig. 3). The technique is labour intensive, and its effectiveness is unquantified. As pink pigeon populations have increased in areas that have been intensively trapped (Jones and Swinnerton 1997), it can be assumed that trapping has achieved some success as an emergency measure. However, this method was introduced as a short-term solution in the absence of any scientific data. It is clear that pink pigeon conservation will require intensive predator management in the foreseeable future. This needs to be as efficient as possible in order to be sustainable in the long-term (Safford and Jones 1998). In order to gather the information needed to make informed management decisions, we studied the ecology of *H. javanicus* on Mauritius from 1997 to 2000. The aim of this paper is to give an overview of the ecological information gained during the study and show how it can be applied to improving current management regimes. We also discuss alternatives to the current method of box-trap control and highlight how certain aspects of the ecology of the species make some alternatives more viable than others.

METHODS

Census techniques

We calculated relative mongoose densities from footprints collected from track stations. The track stations were set up by sieving fine sand on to bare earth in circles with a diameter of 2 m, and scented in the centre using approximately 5 ml of fish oils. These stations were grouped together in sets of four, 10 m apart, and 10 groups running

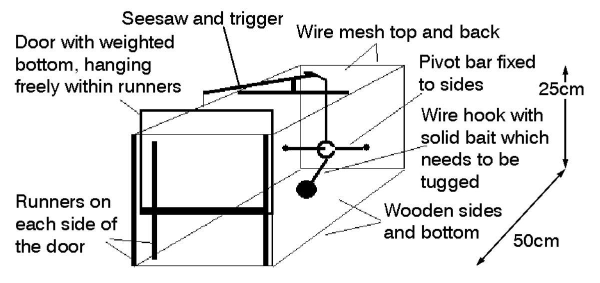

Fig. 2 The design of traps used for mongoose control on Mauritius.

for a length of 1 km constituted a transect. Each transect was at least 5 km from its nearest neighbour, so that animals associated with one transect could not influence the number of visits recorded in another transect during the censusing periods. Transects were run for 10 days, and each day the total number of mongoose visits to each station was recorded. Track stations were re-scented and re-smoothed daily. The daily scores were treated as repeated measures over 10 days when analysed. No attempt was made to identify individuals from prints. In the analyses the total number of visits were used as an indirect measure of abundance. The number of feral cat (*Felis catus*) visits were also recorded, as other authors have found that populations of some mongoose species often interact with those of other carnivores in an ecosystem (Palomares *et al.* 1996). The census technique is an adaptation from those described by Roughton and Sweeny (1982) and Sutherland (1996).

Scent station transects were set up and run seasonally in each of the broad habitat types described by Page and D'Argent (1997) in their vegetation survey of Mauritius. These are described below:-

- Forests which largely consist of native species.
- Forests that are mostly exotic but have some native vegetation in them.
- Wholly exotic vegetation, consisting of mixed scrub, grassland and acacia forest.
- Riparian vegetation, habitats that fall within 20m either side of streams and rivers and around lake edges.
- Plantations of tea, eucalyptus and conifers.
- Sugar cane.
- Coastal vegetation.

Radio-telemetry

Radio-tracking was carried out in Yemen (Fig. 1), a hunting estate in south-west Mauritius. This study site was chosen as it had many different habitat types within it as described above, which would highlight small-scale structural features within habitats to which animals are attracted. 14 mongooses (seven males and seven females) were caught in box traps, transferred to hessian handling sacks and immobilised by intra-muscular injection of 0.2 ml ketamine hydrochloride (Vetalar®). Animals were then collared and tracked. Collared animals were continuously followed for 10 days using the methods outlined by Harris *et al.* (1990). Radio fixes were recorded every 15 minutes, and animals were located to a 25 m by 25 m square on a grid overlaid onto habitat maps of the area. The habitats of the study area were divided into rocky areas, paths, riparian areas, sugar cane and long grass areas (>knee height), short grass areas (<knee height), mature forest, and immature forest/scrub. The habitat groupings are based on qualitative structural similarities between habitat types, for example sugar cane and long grass were grouped. All radio-tracking was carried out between sunrise and sunset, since the species is diurnal (Pimental 1955; Gorman 1979; Coblentz and Coblentz 1985). Minimum convex polygons (MCP) were estimated from the data using the program

CALHOME (Kie *et al.* 1996). Home ranges were then overlaid onto habitat maps created using the GIS package Arcview (ESRI 1996), and compositional analysis was carried out to determine habitat use by the species, as described by Aebischer and Robertson (1993).

Long-term trapping

The predator management regimes at MWF field stations were formalised in 1997. The field station at Combo (Fig. 1) only became operational in 1999 and has not been included in the analyses. The other field stations are described as follows:

- Brise Fer: set up in 1988, this is the longest running MWF field station. It contains the largest tracts of pristine native vegetation and was the site of the first successful pink pigeon releases in the late 1980s. As a result it is also the site with the longest history of predator management. It had 24 traps.
- Pigeon Wood: this is the site where the last remaining wild pink pigeons were found. Predator management regimes began in this site in 1991. It had 23 traps.
- Bel Ombre: this field station was set up in 1994, when the pink pigeons were first introduced to it. It had 22 traps.

The placement of traps follows the general strategy described in Fig. 3. All field stations had a concurrent history of rat control (*Rattus rattus*), although in Bel Ombre rat control ceased after two years.

Traps were allocated to the following categories:

- Access traps: placed near (though not necessarily on) paths, dry riverbeds or other such entry points into pink pigeon breeding areas.
- Perimeter traps: placed in a protective ring around pink pigeon breeding areas, to stop predators entering the predator-free zone.
- Core traps: grid of traps within an area that is intensively managed to create a predator-free zone. These grids were strategically placed to coincide with areas of high pigeon breeding and nesting activity.

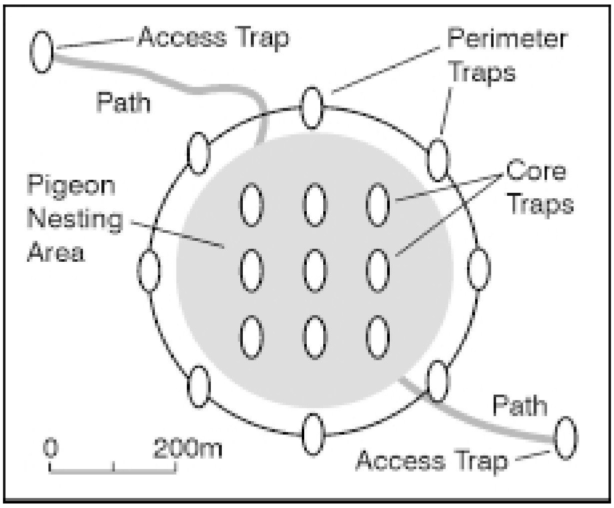

Fig. 3 A stylised trapping grid showing how traps are set out at MWF field stations.

Various habitat features associated with these traps were recorded, and analysed with respect to mongoose trapping success. These were:

- Trap type (Access, Perimeter or Core).
- Main habitat type in a 2 m radius around the trap (rock and bare earth, ground vegetation, understorey or mature tree).
- The main habitat type in a 20 m radius around the trap (scrub and immature forest, guava thicket, mature forest).
- Linear features within a 2 m radius of the trap (none, habitat edge, path and river).

These results were then compared qualitatively to the results from compositional analysis. It should be noted that not all the habitat types recorded in the radio-tracking or census study were represented in the trapping study. This is because the trapping grids were set up as a management measure, whilst the others are part of a study directed solely to understanding the ecology of the mongoose. For example, thickets of guava are frequent in the field stations whilst in places like Yemen they are not. As a result any comparisons drawn are broad and qualitative.

Diet

Carcasses were collected by trapping at Bel Ombre, Brise Fer, and Pigeon Wood, and in the lowland gorges around Mauritius as part of the Mauritius kestrel (*Falco punctatus*) recovery programme from 1984 to 1997. Road kills were also collected opportunistically. Animals captured in box traps were transferred to handling sacks and killed by cervical dislocation. The entire gut was removed and gut contents were washed into a 0.5 mm sieve, and contents were sorted, identified and grouped into six broad categories for statistical analysis. It was assumed that there were no major changes in the different prey groups from 1984 to 1999, the data from different years were pooled to increase sample size, and presented as percentage frequency of occurrence of prey items.

RESULTS

Habitat use

Radio-telemetry

Compositional analysis (Aebischer and Robertson 1993) was used to quantify the habitat use by radio-collared mongooses. This technique compares the number of radio fixes per unit area recorded in each habitat type with each of the other habitat types in turn. The total number of times a habitat is preferred over others is then summed and used to rank that habitat type. Habitat types of equal rank were then grouped together. Habitat use was found to be non-random (Wilks' lambda = 0.77, P<0.001), and then ranked and grouped (shown in bold) as shown below. In this example there are two clear groups; 'A' is the most preferred habitat group while 'B' is not. As a result of this procedure some habitat types do not neatly fit into a particular rank group and have a wide "band", as shown by the "forest" habitat below:

- Rocky areas and riparian habitats (A)
- Mature forest (A, B)
- Scrub, long grass/sugar cane, short grass, paths (B)

Trapping

A generalised linear model (GLIM) was used in Minitab (Ryan *et al*. 1985) to test whether the placement of traps had any significant effect on subsequent captures of mongooses. Traps of different categories (access, core, and perimeter) did not show significantly different capture rates for mongooses (F = 0.24, P = 0.78). They were subsequently pooled in order to increase sample sizes for the rest of the analyses. The habitat type 20m around the trap did significantly influence captures. Figure 4 shows that mongoose capture was higher in forest habitats than in scrub habitats. Capture rates were generally low, with an average of 0.03 mongooses/trap night in each of the field sites of Bel Ombre, Brise Fer, and Pigeon Wood. This was partly due to capture of non-target species, such as tenrecs (*Tenrec eucaudatus*), introduced from Madagascar.

The sex ratio as shown by trapping was male biased, ranging from 3:1 to 5:1. It is assumed that this was not a result of sexual bias in trapability, as data from other trapping studies in Mauritius not presented here show a 1:1 sex ratio. Instead, this is probably caused by immigration into areas where mongooses have been removed by trapping. In many small carnivore species, males show a greater tendency to disperse than females, and the same may be true for mongooses, as reported by other authors (Hoagland *et al.* 1989).

Census

A GLIM model was run in Minitab (Ryan *et al.* 1985) to relate the effects of cat visitation, season, and habitat type on mongoose visitation rates. Interactions between these factors were also tested. A deletion test was carried out where non-significant factors and interactions were removed from the model and the test was repeated until only significant factors and interactions were left. Mongoose visitation varied significantly in different habitats, and in different seasons. There was also a season-habitat interaction suggesting seasonal movement between habitat types ((1) Habitat; DF = 6, F = 55.30, P<0.001 (2) Season; DF = 2, F = 15.82, P<0.001 (3) Season-habitat; DF = 12, F = 5.19, P<0.001). Mongoose visitation rates were highest in degraded forest and riparian habitats, and lowest in coastal habitats and sugar cane. Visits by feral cats had no effect on mongoose visits (DF = 1, F = 0.10, P = 0.753).

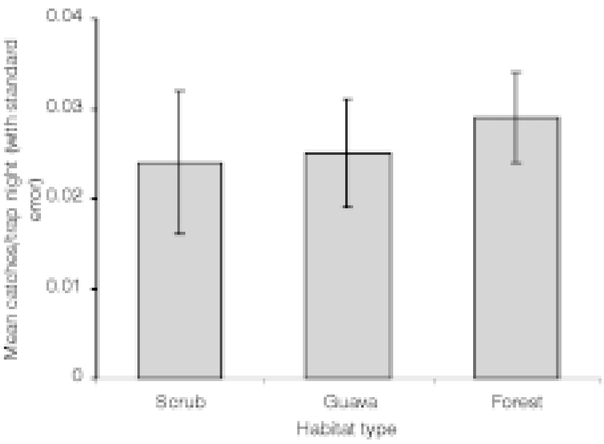

Fig. 4 The effects of habitat in a 20m radius around traps on capture success (GLIM: Habitat type; DF=2, F=2, P<0.05).

Territoriality

Males' and females' ranges did not differ significantly (N = 14, F = 1.48, P = 0.28), although this is based on a small sample size. The mean MCP home range size (± standard error) was 0.77 ± 0.07 km^2 and ranged from 0.25km^2 to 1.10km^2. Density estimates from capture-mark-recapture in these areas ranged from 25.6 to 52.4 animals/km^2 (mean 37.3). Home ranges overlap considerably and the species is not territorial. During other trapping studies on Mauritius individual traps caught up to six different animals (marked with ear tags) within the space of a week. This supports earlier findings that *H. javanicus* is not a territorial species (Gorman 1979). Populations in small areas had rapid turn over, and during trapping studies, animals caught and tagged in one season were not caught in the next trapping session three months later. This is more likely to be a consequence of seasonal movements rather than induced trap-shyness. Evidence to support this is borne out by the fact that animals that had been radio-collared or ear-tagged were caught on several occasions after their initial capture, while the census studies described above have shown that populations show seasonal movements between habitat types. Subsequent modelling (Roy 2001) showed that mongoose populations are more sensitive to changes in survival than to changes in fecundity.

Diet and impact on pink pigeon populations

A total of 458 mongoose guts (365 male and 93 female) were analysed. The frequency of occurrence of all main prey groups is presented in Table 1. Birds were the least

Table 1 The frequency of occurrence of different food types in 458 mongoose guts collected from 1986 to 1999.

Prey type	Frequency of mongoose guts with food type (%)
Rodents and shrews	46
Tenrecs	20
Invertebrates	20
Refuse, carrion, and plant material	18
Reptiles and amphibians	15
Birds	6

frequent of all prey items and only occurred in 6% of guts, while rodents and shrews were the most frequent and occurred in 46% of the guts examined. Despite this low incidence of bird predation we believe that mongooses do have an impact on pigeon populations. To support this a time series multiple regression was carried out using the data from Brise Fer, a site with the longest history of predator management. This analysis showed that pink pigeon mortality correlated significantly negatively, albeit weakly, with increased numbers of mongooses trapped and removed from the period 1996-1999 (r^2 = 18%, P = 0.012) (Roy 2001).

DISCUSSION

The long-term management of mongooses

Mongooses in Mauritius will have to be managed for a long time into the future. Mauritius is too large, and current technology is too limited, for any attempts at eradication of this widespread introduced species. However, larger areas need to be managed for conservation to be more effective. Firstly, larger reserves improve the long-term chances of survival of native species (MacArthur and Wilson 1967). Secondly, populations of animals, in particular carnivores, are more predictable in larger areas rather than smaller areas, making their management easier (Smallwood 1999). Therefore the management of mongooses over larger areas should be a primary goal in the future. Control of other invasives often involves techniques that have been developed by adapting existing knowledge from gamekeeping or agricultural protocols. This is illustrated by the stoat (*Mustela erminea*); many of the control methods and equipment used for managing this species in New Zealand are based on European gamekeeping techniques (Reynolds and Tapper 1996). Mongooses, however, have very little long-term cultural history of control. In order to control mongooses on islands, the favoured way is trapping, mostly using Tomahawk or box traps (Coblentz and Coblentz 1985). These techniques are too labour intensive for use over large areas, but they can be improved if we take into consideration some of the findings from the study. The improvements are presented as follows:

Trap spacing

The smallest home range was found to be 0.25 km^2 so it is recommended that the minimum trap density should be one trap every 0.25 km^2. However, home ranges were seen to overlap greatly and animals were not seen to be territorial, so a greater trap density than this is preferable. Further experimentation with trap spacing and different bait types is needed to show if the trapping biases discussed earlier are a true indication of the sex ratio of the mongoose population or an artefact of trap shyness of females. Placing more than one trap at each trap site should improve the number of captures a night. This is because currently traps can only catch one animal a night and often traps become temporarily inoperative due to capture of non-target animals such as tenrecs. As mongooses are not territorial, it can be assumed that there is a likelihood of more than one animal being present over a small area at any one time.

Trap siting

Forested habitats feature highly in the habitat preferences of mongooses as shown by radio tracking. Long-term trapping studies also show that traps in forested habitats are more successful, while census studies show that the highest densities of mongooses occur in degraded forest. It is clear that control should focus on this habitat type, especially where resources are limited.

Predator control in a multi-predator environment

Only 6% of mongoose guts had bird remains in them. A low level of predation can still affect the long-term survival of pink pigeon populations. Predator control should be concentrated in areas frequented by species of conservation concern. It should also be noted that 46% of mongoose guts had rodent and shrew remains in them. Controlling just mongooses on their own may release rats from predation pressure, and may also release feral cats from feeding competition. As cats and rats are also major bird predators, mongoose trapping should be carried out as part of a regime that targets all of the predator species together. Further research on the interaction between the different predator species is required to allocate resources optimally between the different predator species.

Alternative control measures that could be adopted in future

Long-term predator control over larger areas is possible if we adopt alternative, less labour-intensive approaches to control, as it is clear that this is not possible by trapping alone. These alternatives could be used for large-scale management, possibly combined with small-scale intensive trapping regimes at particularly vulnerable locations. The ecology of mongooses on Mauritius, and of species of conservation concern, favours some control methods over others. These are discussed below:

Non-lethal physical techniques

Barriers: Fencing and placing metal barriers on trees may prevent mongooses from reaching sensitive areas or nests respectively as the species does not climb very well. Fencing may not be an option, however, as it is a costly technique and would require a lot of maintenance in a country like Mauritius which frequently suffers cyclones. Also, Mauritius has other introduced predators that do climb well, such as the macaque (*Macaca fascicularis*), and such barriers would not prevent predation by these species. If fencing is to be used, the construction should prevent predation by all major introduced predators, especially as it is so costly. Fencing is also an inflexible approach to predator management. Once erected, it would not be easy to respond to changes in the boundaries of conservation management areas through regeneration and recovery; a scenario which is likely to occur on Mauritius as pink pigeon populations expand as the forest recovers. Fencing is not a viable alternative at its current cost. Should the cost decline in future, sensitive areas may be eradicated of ground-based predators by trapping or poisoning, fenced, and ingress of predators prevented by monitoring and preventative measures around the boundaries to help prevent reentry into enclosed areas.

Habitat management: Hitherto the promotion of unfavourable habitats to reduce mongoose populations or create unfavourable hunting conditions has had mixed results (Alterio *et al.* 1998). Radio-collared mongooses spent less time in open areas of short grassland than in any other habitat and they may also be den-site limited (Roy 2001). So removing denser ground vegetation in pink pigeon feeding areas and removing potential den sites may be beneficial in combination with other management practices.

Non-lethal chemical techniques

Fertility control: Reducing the fertility of mongooses via baits placed in the field is not an option as the technology has yet to be developed. Also, the pink pigeon populations around Mauritius are at a stage where any short-term predation is likely to affect the survival of the population and sterile mongooses could still prey upon pink pigeons in the short term. There would also be no guarantee of reaching every animal, especially as immigration rates of fertile animals are potentially quite high in some habitat types.

Conditioned taste aversion: By placing chemically-altered foul-tasting prey items in the environment, predators can sometimes be "trained" to avoid that particular prey type. The technique has been used to reduce fox predation on the eggs of ground-nesting birds (Conover 1990). However, mongooses are not the ideal predator for this technique as they are short-lived with a high population turnover through recruitment and immigration. There is a constant influx of "untrained newcomers" into the environment and each animal has only a limited exposure to the foul-tasting bait. This is not a viable alternative to current control methods.

Lethal methods

Alternative trap types and baits: There are many alternative trap types available on the market, such as Fenn traps, but the costs of these far exceed those of the home-made traps employed in Mauritius. Using them would greatly add to the expense of control operations, making them unsustainable in the long term. Also, the traps currently used present no danger to native birds and are easy to operate by volunteers with minimal training. This is not the case with some of the other traps available on the market. The current trap type can be improved, however, by making them from standardised parts that are easy to assemble and dismantle so that field operators can replace trap parts rather than whole traps as part of their maintenance.

Poisoning: Mongooses are poisoned successfully in Hawaii with diphacinone (Lindsay and Mosher 1994; Dusek and Aeder 1995), a poison to which they appear to be very susceptible. Baits placed in the environment have a high probability of being taken as *H. javanicus* has been shown to take baits readily (Creekmore *et al.* 1994).

Of all the alternatives outlined, poisoning is the most viable in the near future. However, poisoning campaigns often affect non-target species, and if the dosage of poisons within bait formulations is species specific, large non-target species such as cats may not be killed quickly and humanely, raising ethical issues. Poisoning campaigns can be made more species specific using the "Achilles' heel" approach (C. Marks pers. comm.). This approach uses physiological, ecological and behavioural attributes unique to a species to improve the species specificity of a poisoning campaign, for example by placing poison baits in rock crevices the baits can be made more accessible to mongooses and less accessible to feral pigs or monkeys. This approach is already being used to develop baiting campaigns for foxes (Busana *et al.* 1998) and feral cats (Fisher *et al.* 1999) in Australia. It increases efficiency and minimises bait uptake by native marsupial carnivores (Belcher 1998), and field trials using bait markers, which show up in the whiskers or the blood of captured animals, are being used to evaluate the specificity of bait delivery methods (Fisher *et al.* 1999). Similar poisoning campaigns for mongooses would need to exploit behaviours unique to mongooses to reduce uptake by non-target species. For example, typical mongoose den sites could be targeted with bait delivery stations. However, for generalist, opportunistic species like mongooses, there are few "unique" food preferences that could be exploited to prevent uptake by opportunistic non-target species such as insects, which could then be eaten by native insectivorous birds like the merle (*Hypsepetes olivaceous*). With a better understanding of its autecology, it may be possible to identify the mongooses' "Achilles' heel". This will facilitate the development of efficient, species-specific control programmes.

An alternative to the Achilles' heel approach is to formulate poisons, or use mixed baits so that they quickly kill multiple target species (i.e. cats, rats and mongooses). Controlling multiple predators may be a better control strategy where there is a risk of meso-predator release as a result of single species control. The situation on Mauritius also favours this approach because, unlike Australia, there are no native mammals that could be affected by toxicants in poison baits, and all three predators are believed to have a significant impact on native species.

The future of mongoose control in Mauritius; is eradication ever possible?

Mauritius has an area of approximately 2000 km^2 and is densely populated with a population of 1.5 million people. It has a large mongoose population throughout the island, some areas of which are very mountainous and relatively inaccessible. Mongooses may be controlled or even eradicated in the distant future with the use of aircraft to drop poison bait formulations, along with the co-ordinated efforts of different government departments and authorities, but this is unlikely in the near future due to lack of resources. Further study is required to test poison bait uptake in the field using bait-marking studies similar to those described earlier. Also bait formulations need to be optimised to ensure that they do not pose a risk to non-target species, before poisons are broadcast over large areas.

In the short term mongooses can be more efficiently controlled or even eradicated over small, sensitive areas with conservation value using trapping in combination with poisoning. In conjunction with this, mongooses can be controlled over large buffer areas by poisoning. This would achieve control over a large area and reduce immigration into sensitive core areas. The poison could be delivered using bait delivery stations or it could be placed by hand at bait points as long as safety issues are addressed and field trials on bait uptake are carried out.

Any improvements in ecological management of *H. javanicus* will require a greater understanding of its ecology, and this requires more information on the population and behavioural ecology of the species in both its native and introduced range. In particular, we need to investigate the uptake of baits in the field and the interactions between mongooses and other predator species. If management regimes can be made more cost-effective and efficient, in time larger areas could be managed. The work on Mauritius carried out so far adds to our knowledge of the species, and can be applied to other islands where the species has not been studied.

In this respect, Mauritius is an ideal site for such a study. It is a typical, tropical oceanic island with all of the problems faced by other islands, such as habitat loss and introduced species. It is a biodiversity hotspot, and much of its flora and fauna is well studied and has been well recorded in history. Very simple conservation techniques have been used to save some of the world's rarest species, like the Mauritius kestrel, which has been restored to a state where the population is self-sustaining or requires minimal management (Jones *et al.* 1995). Mongoose control played a significant role in conserving the Mauritius kestrel. Conservation techniques are easier to develop on Mauritius than on other islands where the fauna and flora or the geography have not been as well studied. Techniques developed on Mauritius can in future be adapted and applied to other, lesser-studied island ecosystems, where other conservation issues need to be addressed.

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Eradication of feral pigs (*Sus scrofa*) on Santa Catalina Island, California, USA

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Abstract Control efforts initially designed to reduce feral pig numbers, and subsequently altered to remove all feral pigs began on Santa Catalina Island, California, in 1990. The programme occurred in four phases, each with different management objectives and geographical emphases. Phase I involved reducing pig numbers on the western 20% of the island. Control efforts were expanded in Phase II to include reduction of pig numbers island-wide. Phase III involved eradication of pigs on the west end with continued control elsewhere. Phase IV involves the removal of all pigs from the island and has a scheduled completion date of 2004. To date, at least 11,855 pigs have been killed. Removal strategies include trapping (39%), systematic hunts with dogs (30%), systematic and opportunistic ground hunts (26%), aerial hunting (3%), and night spotlighting (2%). Catalina has a resident human population of 5000, and more than 1,000,000 visitors annually, requiring close integration of removal efforts with other island activities. At least 96,500 hours have been expended by staff, contractors, and volunteers. Total costs exceed USD3,175,000. Less than 300 pigs are estimated to remain in October 2001.

Keywords Santa Catalina Island; feral pigs, *Sus scrofa*; eradication strategies.

INTRODUCTION

Attempts to control or eradicate feral pig (*Sus scrofa*) populations on islands have been undertaken in different parts of the world for many years (Hone and Stone 1989; Katahira *et al*. 1992; Lombardo and Faulkner 2000). Since 1990, intensive efforts to remove pigs have occurred on Santa Catalina Island, the third largest of the eight California Channel islands. The programme has evolved through the last decade with several distinct phases, each with different objectives and on-island geographical emphases (Table 1). Current objectives call for the removal of all pigs from the island. This paper discusses all pig removal efforts since November 1990.

Pigs are not being removed from Catalina because they are non-native, but because of their impact on the island's ecosystem. Near the beginning of the programme a series of vegetation transects were placed on the west end of the island and baseline data collected (Laughrin *et al.* 1994). Results of this and subsequent monitoring indicated an increase in the percent of vegetative ground cover and an increase in species diversity from 1990 to 2000. While many of the species contributing to this increased diversity were exotic plants, the increase in ground cover was considered a positive step as it decreased the potential for continued topsoil erosion. More systematic and complete monitoring programmes for both vegetation and fauna are being established and will provide results in the coming years. However, since both pigs and goats were removed from this area at the same time, making an accurate assessment of recovery due solely to pig removal will be problematic (Gay 1999; Kraus 2000).

Table 1 Phases and objectives of pig removal efforts on Santa Catalina Island since 1990.

Phase	Date	Objective	Location
I	11/1990-4/1991	As many pigs as possible	West End
II	2/1992-6/1996	As many pigs as possible	Entire island
III	7/1996-12/1997	All pigs	West End
	7/1996-6/1998	As many pigs as possible	Eastern 80%
IV	7/1998-present	All pigs	Entire island

Study area

Santa Catalina Island (SCI), lies 32 kilometres (km) south of Point Vicentes, Los Angeles, California, U.S.A (Fig. 1). Ranging in width from <1 to13 km, the 194 km^2 island is approximately 35 km long with a highest elevation of 640 metres. The rugged, mountainous topography of SCI

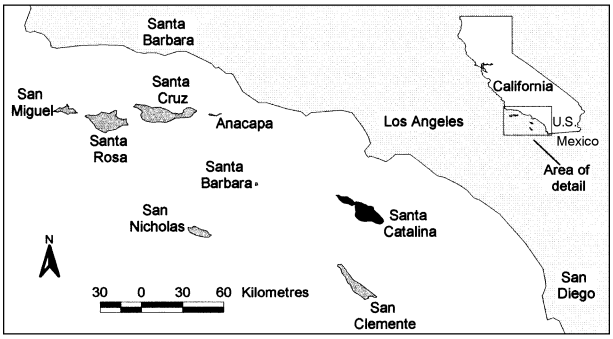

Fig. 1 Santa Catalina Island, the third largest and most populated of the eight California Channel Islands, lies south-west of the large Los Angeles metropolitan area.

Pages 274-286 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

is dominated by a north-west to south-east mountain range containing complex arrays of lateral canyons. Slopes normally range between 20° and 30° although they may be nearly twice as steep. The western 20% of the island (38 km^2) is isolated from the remaining portion by a narrow, low (<15 m elevation) isthmus providing a natural barrier for use in control programmes (Fig. 2). The climate is Mediterranean with wet winters, long dry summers, and relatively mild year-round temperatures. Rainfall varies both annually and spatially over the island but generally averages between 200 mm and 400 mm. At least 14 distinct plant communities have been documented on SCI (Thorne 1967; D. Knapp pers. comm.). Grassland accounts for nearly 30% of the total area, while chaparral and mixed-oak woodland comprises 44% of the vegetative cover. Twenty percent of the island is covered by low-growing coastal sage scrub, maritime desert scrub or *Opuntia* scrub and the remaining 6% is eroded badlands, cultivated areas, or developed sites (Minnich 1980). A depauperate native mammal fauna includes island fox *(Urocyon littoralis catalinae*), Beechey's ground squirrel (*Spermophilus beecheyi nesioticus*), deer mouse *(Peromyscus maniculatus catalinae*), harvest mouse *(Reithrodontymis megalotis catalinae*), ornate shrew (*Sorex ornatus willetii*), and at least five bat species (Von Bloker 1967). All terrestrial mammal species, excluding bats, are endemic to Catalina at the subspecies level. Introduced alien herbivores currently include feral goats (*Capra hircus*), American bison *(Bison bison*), mule deer *(Odocoileus hemionu*s), feral pigs, and a small herd of black buck (*Antelope cervicapra*). Numerous sheep *(Ovis arie*s) were removed from the island by the mid-1920s and the majority of cattle (*Bos taurus*) were gone by the late 1950s (O'Malley 1994). Thirty-nine bird species breed on the island and more than 225 species have been recorded (Schoenherr *et al.* 1999; R. Hansen pers. comm.). There are 11 species of reptiles and amphibians regularly recorded on the island. There are 472 native plant species and 235 introduced plant species. Of the native species, six are restricted to just Santa Catalina Island while at least another 22 are restricted to the California Channel islands and are not found on the mainland.

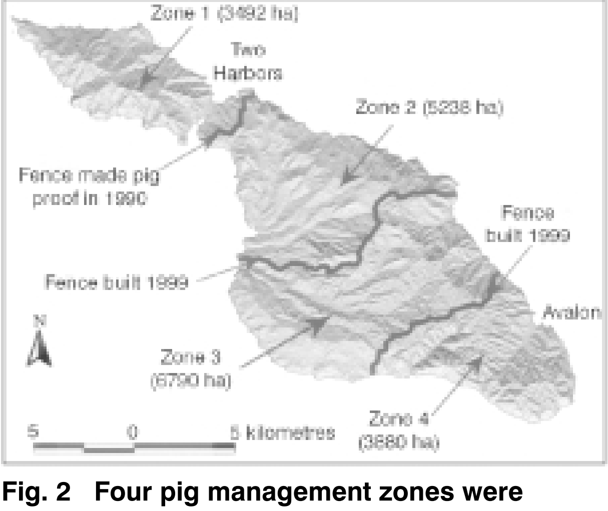

Fig. 2 Four pig management zones were created on Santa Catalina Island by constructing 29 km of fence.

In addition to native plant, vertebrate, and invertebrate species, Catalina also has a resident human population of approximately 5000 people and nearly 1,000,000 visitors per year. The town of Avalon has 4500 residents and receives the majority of the visitors. The remaining population is located in the small town of Two Harbors (pop. 200) or at various camps and facilities. The Santa Catalina Island Conservancy (Conservancy) owns and manages 88% of the island with a primary goal of natural resource protection while still allowing appropriate public access. Members of the William Wrigley Jr. family established the Conservancy as a private, non-profit organisation in 1972. The Santa Catalina Island Company, which operates most of the island's commercial ventures, owns 11% of the island and allows the Conservancy access to its land for a variety of conservation purposes. The remaining 1% is privately or publicly owned.

History and impacts of pigs

During the 1930s, pigs were first introduced to Santa Catalina Island, allegedly for either sport hunting or to help control Catalina's rattlesnake population (Overholt and Sargent 1971). The animals came from Santa Rosa Island where they were originally introduced from an unknown source, most likely in the mid-1800s (Collins 1981). During the 1990s, the National Park Service eliminated pigs from Santa Rosa Island (Lombardo and Faulkner 2000). A syndactyl breed of pig was introduced to Catalina prior to 1955, although only a very few pigs with this morphological characteristic are still sighted (K. Ryan pers. comm.). The number and distribution of Catalina's pigs vary dramatically year to year. In 1980, the pig population was estimated to be between 1260 and 2040 animals (Baber and Coblentz 1986). Although present over the entire island, densities of pigs are highest in moist canyon bottoms and lowest on exposed ridges. Condition of the pigs is closely tied to the availability of acorns and other food sources. During years with poor quality or quantity of forage, the pig population decreases and animals can be nearly too weak to stand. The presence of antibodies for pseudorabies virus, San Miguel Sea Lion Virus (a calicivirus of marine mammals similar to swine virus), and trichinosis has been documented for Catalina's pigs, indicating prior exposure to these diseases (Timm *et al*. 1994).

The ecological effects of feral pigs on island resources are known from numerous places around the world, although documentation is not always easy to obtain (Wood and Barrett 1979; Stone 1985; Coblentz and Baber 1987; Pavlov *et al.* 1992). Effects may vary depending upon the density of pigs and relative sensitivity of the ecosystems (Bratton 1975; Singer 1981). The impacts of feral pigs on the ecosystems of Catalina and the other California Channel Islands are documented (Thorne 1967; Hochberg *et al*. 1980; Baber 1985; Baber and Coblentz 1986; Sterner 1990; Peart *et al*. 1994; Lombardo and Faulkner 2000). Impacts on Catalina include extensive rooting of slopes, disturbance of soils around the base of native trees, including the endemic genus of Catalina Island ironwood trees (*Lyonothamnus floribundus* spp. *floribundus*), uprooting and trampling of small seedlings and saplings, and direct consumption of small native vertebrates and invertebrates (Baber 1985; Garcelon 1995, 1998). The decline in range and population size of several native plant species is suspected to be associated with pig presence (Thorne 1967). Pig foraging on acorns is likely a major factor in the almost complete lack of regeneration of two endemic oak species (*Quercus pacifica* and *Q. tomentella*). As noted by Lombardo & Faulkner (2000), areas of pig rooting create optimum growing conditions for invasive exotic plants such as spiny clotbur (*Xanthium spinosum*), fennel (*Foeniculum vulgare*), and horehound (*Marrubium vulgare*). On other California Channel Islands, archaeological sites, especially those located in caves, have been heavily impacted by feral pig rooting and bedding behaviour (Lombardo and Faulkner 2000). Given the widespread evidence of Native American habitation on Catalina in comparable sites, we think it is reasonable to expect similar degradation to the island's cultural sites.

Previous control efforts

Since their introduction, island residents, land managers and visitors have hunted pigs for both sport and subsistence. Prior to 1990, the goals of feral pig management programs on Catalina were not well defined and involved intermittently reducing pig numbers, utilising on-island resources and available time. Although sport and subsistence hunting was widespread, and land managers shot pigs opportunistically, it is unlikely, given the reproductive capacity of pigs, that this level of hunting had significant island-wide effect on pig numbers. This finding is consistent with preliminary results of the effectiveness of sport hunting in reducing feral pig populations in Hawaii and New Zealand (Barrett and Stone 1983; Clarke 1988). Until 1990, pig numbers were more likely influenced by the amount of annual rainfall, the acorn mast crop and the availability of other food sources (D. Propst pers. comm.).

Current control programme

Although Catalina faces many natural resource challenges and issues, the removal of non-native animals, particularly pigs and goats, has long been identified as a top priority. Bruce Coblentz (pers. comm.) raised the possibility of goat eradication in 1973 and again in 1980. Finally, in the late 1980s, the administration and board of the Conservancy, with strong urging from Dr. Robert Thorne, director of the Rancho Santa Ana Botanic Garden, discussed increasing feral animal control efforts (D. Propst pers. comm.). Goats were the initial focus of removal efforts, however, the control of both species became top priorities by 1990.

Although almost the entire island is privately owned, pigs are considered the property of the State of California and, as such, all management programmes must be authorised by the California Department of Fish and Game (CDFG). In 1990, a Memorandum of Understanding (MOU) between the CDFG, the Conservancy, and the Institute for Wildlife Studies (IWS), was signed to allow the Conservancy to eradicate pigs on the west end of Catalina. In 1992, a new MOU was signed allowing the Conservancy to eradicate pigs over the entire island. The Institute for Wildlife Studies, a non-profit conservation group, was a signatory to the MOUs as they were contracted by the Conservancy to implement the control programmes. Both MOUs allowed pig carcasses to be left on site rather than utilising or burying them as is normally required in a depredation permit. This permission was crucial to the success of the programme due to Catalina's rugged, inaccessible terrain, the inability to follow government standards for preparing meat for human consumption, and the logistical constraints of obtaining approved butchering and transport facilities.

METHODS

Planning

Many methods of feral pig control have been tried and evaluated throughout the world (Hone and Atkinson 1983; Breuer 1987; Davis 1987; Hone and Stone 1989; Sterner 1991; Sterner and Barrett 1991; Barrett and Birmingham 1994; Jenkins *et al*. 1994; Choquenot *et al*. 1996; Lombardo and Faulkner 2000). Catalina's pig control programme should be viewed as an adaptive management program. Each phase of the 10 year programme provided new information which was incorporated into subsequent phases to improve results and efficiency. Methods were refined and adapted throughout the programme to meet new challenges and to help the program fit in with other activities occurring simultaneously on Catalina. With a resident human population and a million visitors per year, some methods could not be utilised on Catalina that might otherwise have proved more efficient in terms of efficiency, results, and cost. Both humaneness and the ability to accomplish programme goals were considered when deciding on appropriate methods.

Removal techniques

Phase I

The western end of the island was chosen as the first area to eliminate as many pigs as possible, due to its relative isolation provided by the isthmus barrier. In addition, an existing 5 kilometre bison fence on the east side of the isthmus was made pig-proof during the early stages of Phase I (Fig. 2). Phases I, II, and III utilised the two management areas created by this fence. Evaluating different techniques for potential use in reducing pig numbers island-wide was a second goal of Phase I.

The following combination of control techniques was used in Phase I: trapping, ground hunting without dogs, ground hunting with dogs, and aerial hunting by helicopter. Each method has advantages depending on season and localised pig density. In addition, limited sport hunting by archers and poaching took place during this period. Control of feral goats was conducted at the same time.

Two trap types were utilised. The majority were all medium-gauge chain link, 1 m x 3 m box traps with a drop door triggered by a line rigged over the bait located at the rear of the trap. The second type, also a box trap, had solid bottoms and a different drop-door trigger mechanism. Potential trap sites along roadsides were pre-baited up to one week prior to placing the trap. Bait trails from 0.1 to 1 km extended from the trap sites to attract pigs in from remote areas. Due to good availability and ease of handling (23 kg sacks), commercial pelleted pig finishing feed was chosen as bait. In addition, any seeds in the feed were sterile thus ensuring no new plants would be introduced to the island. Baited sites were monitored and replenished with bait as needed every other day. One or two traps were then placed at sites showing the highest bait consumption. Traps were set each evening for seven consecutive days and checked early each morning. Pigs found outside the traps were shot with rifles while those trapped inside were dispatched with a .357 magnum pistol. All animals were aged and sexed. Estimated weight and female reproductive status were also noted (Sterner 1991).

Opportunistic ground hunting occurred whenever pigs were sighted. Two ground-hunting trips with dogs, using three Catahoula and two Plott hounds, were conducted. Two to three hunters worked with three to four dogs. Dogs would locate, track, and detain a pig until a hunter came to dispatch it. Limited aerial hunting of pigs occurred during one of the goat removal aerial hunting trips. Phase I ended when island staff departed for other positions and were not replaced.

Phase II

Ten months passed before control efforts were resumed in Phase II. Financial constraints, the opinion of Conservancy administration and board members that pig numbers were sufficiently reduced on the west end to promote recovery, and the uncertainty on the part of some board members whether eradication could be achieved, were all reasons contributing to the delay in resuming the hunting efforts. When the decision was made to renew control efforts, the focus shifted to the entire island rather than just the west end. Financial constraints were again a factor in pursuing a control programme rather than an eradication project.

Phase II methods primarily followed those established in Phase I except trap design was changed to a 1 x 1 x 2 m box trap constructed from pipe, and a corral trap was added utilising 2.5 m panels that could be built in a variety of shapes. The size and design of the corral trap, coupled with adequate pre-baiting and good placement, allowed multiple captures of pigs in a single trap. Although the door was counter-weighted to allow additional pigs to enter, most multiple captures were a function of the larger trap size allowing several animals to enter before the door closed. The number of traps used was increased each year until a total of 23 corral traps and nine box traps were being utilised. In addition to daytime ground hunting without dogs, spotlighting was also instituted. Using a large spotlight, hunters would patrol roads after dark until an animal was seen, at which time rifles with laser sights were used to dispatch the pig. Staff from IWS acquired their own dogs, and used them more extensively than in Phase I. Dog breeds used in this programme included mostly Catahoula hounds, Plott hounds and crosses of the two breeds. All dogs were trained using shock collars to avoid non-target species. All dogs were vaccinated, and any injuries were immediately treated by project staff or a veterinarian. Aerial hunting by helicopter was again used, including missions flown at dawn or dusk using a Forward Looking Infrared (FLIR) device which has been tried elsewhere with varying success (Lombardo and Faulkner 2000). FLIR technology, which measures differences in temperature, detects warm-blooded organisms as they stand out from the surrounding cooler physical environment. Staffing during this phase normally consisted of one full-time hunter with occasional seasonal help, especially during the time when blood samples were being obtained for the disease study (see below).

In addition to recording standard age, sex and reproductive status of shot pigs, a subjective assessment of the pig's health condition was noted starting in 1992. In 1993, blood samples were collected during this phase which established the presence of pseudorabies virus and San Miguel Sea Lion Virus antibodies in the pig population (Timm *et al.* 1994). The likely exposure of island pigs to these diseases made the U.S. and California Departments of Agriculture, as well as CDFG, unwilling to consider live removal as a viable option (Gonzales 2000). This fact became more important as public awareness of the project increased and animal rights/welfare groups raised questions as to why live trapping and relocation were not tried.

Phase III

After three years of reducing pig numbers island-wide in Phase II, it became clear that control efforts, while effective in reducing numbers, were not resulting in any long-term population declines. Increased efforts were needed to eliminate pigs from the entire island. Before authorising an island-wide removal effort, the Conservancy's Board of Directors wanted to ensure it could be accomplished in a smaller area. In May 1995, IWS submitted a proposal outlining a programme to eradicate pigs from the west end (Zone 1) over a two-year period (Garcelon 1995). Control efforts would continue on the remaining eastern 80% of the island but at a reduced level. By utilising the isolated west end as a test area, IWS would be able to develop an estimate for the amount of effort and cost needed for an island-wide eradication programme.

Phase III methods generally followed those established in Phase II with the following additions. Full-time staff was increased to three; two for the west end eradication work and one for the control work on the rest of the island. The number of corral traps was increased to 32 and the design changed to improve portability with prefabricated chain link panels that could be easily assembled in the field. Helicopters were used opportunistically for aerial hunting when brought to the island for other projects. They were also used to transport bait and traps to remote sites. Pigs that came to baited sites at night were dispatched using rifles equipped with night vision scopes.

Phase III was started in July 1996 and originally was estimated to take two years. After only 18 months, no pigs were known to remain and in April 1998, IWS submitted an island-wide pig eradication proposal (Garcelon 1998).

Phase IV

Phase IV started in July 1998 with an estimated completion time of 4-5 years. Prior to approving the initiation of Phase IV, the Conservancy board made sure adequate funding was either in place or at least committed to cover the duration of the removal programme. Phase IV methods continued to expand on those established in prior phases, with the exception of no aerial hunting. This decision was based on the negative reaction of the general public to use helicopters to shoot feral animals. Although effective, aerial hunting was only a small percentage of the total pig removal efforts. To meet the objective of removal of all pigs from the island, IWS staffing increased to seven full-time employees, and additional housing, vehicles, equipment, and dogs were obtained. An additional 125 corral traps were ordered. A permanent kennel, complete with a septic system, was constructed in the middle of the island with facilities for up to 36 dogs.

As outlined in Garcelon's 1998 proposal, building new fences to compartmentalise the pig population into six zones was part of the planned removal effort. Although technically and logistically feasible to work in larger zones, the need to allow safe and regular access to many parts of the island and to minimise impacts on existing resident and visitor activities required smaller areas. There was much discussion and debate regarding the size of management units and associated fence construction design and costs. Other pig eradication programmes have faced similar concerns and some, such as the programme on Santa Rosa Island, opted to do without fences (Lombardo and Faulkner 2000). However, to maintain other island activities, and to keep previously controlled areas free of pigs in case of an unforeseen slowdown or temporary stoppage of the programme, the Conservancy built 24 km of new fence, creating four rather than six management zones (Fig. 2). The fence cost USD825,000.

The fences are constructed out of 1 m tall hog mesh with a strand of predator barbed wire stretched tautly along the contours of the ground to discourage digging. Although not needed to prevent pig crossing, three strands of barbed wire were added above the hog mesh to make the fence more visible to the free-ranging non-native bison and deer herds. To be effective, the fence needed to be checked regularly to locate any breaks. A programme of regular fence monitoring and repair was designed and instituted using local community volunteers.

The plan for Phase IV called for a systematic removal effort in Zones 2, 3, and 4, as well as continued monitoring in Zone 1. Zones were to be cleared to a point where no or very few animals were known to be left before focusing on the next zone. Trapping along both sides of a zone fence was conducted to reduce pig pressure on the fence. This also created a buffer area and reduced the probability of immigration by pigs into a cleared area in the event of a break in the fence. Even as work in a new zone commenced, the old zone would be monitored regularly and any animal spotted would be immediately removed. Once Zone 2 was completed, a strategic decision was made to clear Zone 4 before Zone 3 in case of unforeseen contingencies requiring extra time due to the presence of the town of Avalon and its numerous human activities, and the potential for public relations problems. The plan for Zone 4 called for trapping the entire area outside of Avalon Canyon during the summer months and then moving into the canyon area immediately surrounding the town with traps and dogs during fall and winter months when visitor activities were at their minimum (Fig. 3). By starting at the outer edges of Zone 4, we hoped any pigs that normally moved in and out of Avalon canyon would be caught outside, thereby reducing the number of pigs needed to be taken near town. Zone 4 eradication efforts began in May 2000 and were slowing down by June 2001. Efforts then shifted to Zone 3, which is projected to be completed by spring 2002. A minimum of two years to find and remove the last few known pigs is then expected. Following that, regular and systematic monitoring will be conducted for another 2-3 years before declaring the island to be pig free.

Although the same methods were employed in Zone 4 as in the other zones, additional measures were taken to ad-

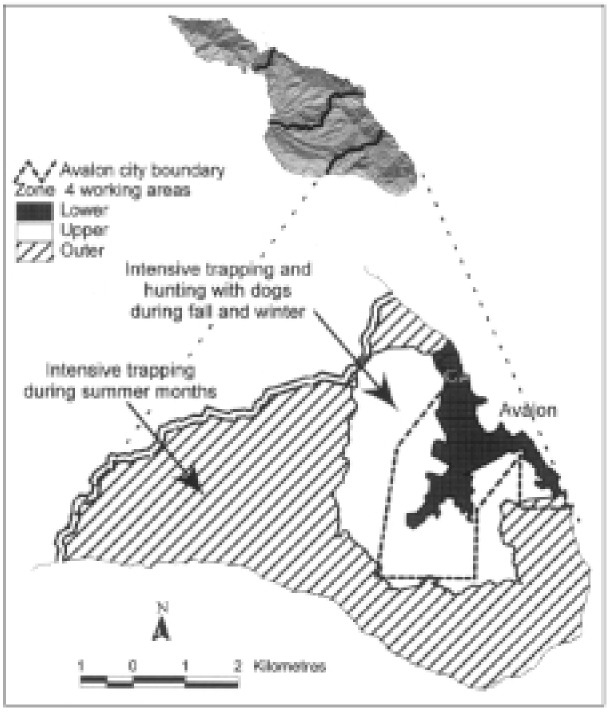

Fig. 3 Detailed map of Zone 4 pig management area, including the town of Avalon, Santa Catalina Island.

dress public safety and public relations concerns. These are addressed in the Public Relations/Education section below.

Monitoring and evaluation

During Phases I and II, when the goal was only to control pig numbers, monitoring of pig distribution and abundance was not considered a critical component of the programme. Staff were limited and the objective was to remove as many pigs as possible without a defined endpoint. However, during Phases III and IV, when eradication was the objective, monitoring changes in pig numbers in order to evaluate success became an important programme component. No systematic permanent transects were established, but staff regularly scouted for sign along game trails, at watering holes, stream crossings, and on dirt roads. In addition, dogs were repeatedly taken into areas when pig densities became low to see if they detected fresh scent. During the winter, disturbance of the new green vegetative groundcover was a good indicator of pig rooting. During summer months, dirt roads and trails were brushed to help detect new pig tracks or, on hard-pan soils, a covering of fine soil was laid down to help detect tracks.

To assist in increasing our understanding of the spatial relationship of animals removed, and to help relate habitat and topographic features with animal densities, the island was overlain with a 500 m^2 grid and each grid given a unique alphanumeric code. All trap sites and each pig removed were then located on the grid and entered into a geographic information system (GIS) database. As more pigs were removed, GIS analysis provided insight into where we might expect higher pig densities as we moved into new zones. As numbers of pigs diminished and animals were only found in groups of one or two, regular systematic monitoring was instituted. At this point, it was very important to receive sighting information from the general public and Conservancy staff, as locating animals became the critical time-consuming task.

Public relations and education

An often-overlooked, but increasingly-necessary component of animal removal programmes is working with or at least informing the general public of progress and plans (Sterner 1991; Barker 1995). For Phases I, II, and III there was very little public interaction or concern relating to the programme. Generally, any dialogue occurred on a one-to-one basis. As the programme scaled up, its visibility increased and the Conservancy felt the need to provide public information on the reasons and scope of the project. This was particularly true as we implemented trapping and hunting in and around the town of Avalon.

In 1999, the Conservancy held a series of public community forums to explain the scope of the eradication project and to answer questions. The forums brought forth numerous questions and concerns, although most of the concerns focused on the goat rather than the pig removal programme. However, realising the potential for impacts on both residents and visitors when control activities began in the Avalon area, the Conservancy began meeting with invited city and county agencies and affected business interests nearly a year and a half prior to actually starting work in Avalon Canyon. At these meetings, we outlined our approach and asked for input from all parties, particularly regarding safety and logistical issues. As a result the following measures were taken prior to or during work in Zone 4: (1) all trail heads in the area were posted with signs in both English and Spanish outlining the nature of the program and who to contact for additional information, (2) all traps in Avalon canyon were posted with bilingual signs describing their purpose, (3) all traps in Avalon canyon were locked open when not actually set to capture pigs, (4) all pigs caught in traps were chemically immobilised and removed rather than being shot in the trap, (5) all pig carcasses were removed from the Avalon Canyon area, (6) the sheriff's department was notified each and every day IWS was in the area with a potential to discharge firearms, (7) any affected businesses, such as the horse stable and trail riding operations, were notified before IWS worked in their areas, and (8) a mailing in both English and Spanish was sent to all island post office box holders (there is no home mail delivery on Catalina) outlining the programme and who to contact for additional information. The Conservancy also submitted a number of articles to the local newspapers, mentioned the removal programme at all regularly-scheduled Conservancy education and field activities, and made staff available to speak to any local organisations or homeowner groups requesting additional information.

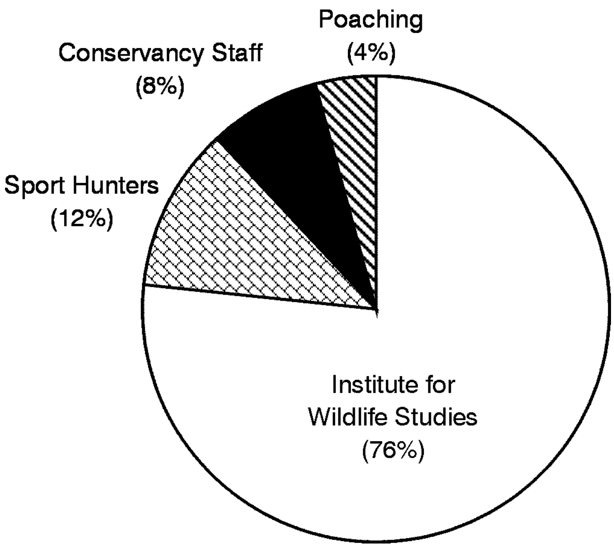

Fig. 4 Percentage of pigs removed by different personnel between 1990 and September 2001 on Santa Catalina Island (n = 11,855). Pig numbers estimated for staff and poaching percentages. Numbers for sport hunting calculated by dividing known number of sport hunter days per year by estimated take ratio. IWS numbers came from detailed records.

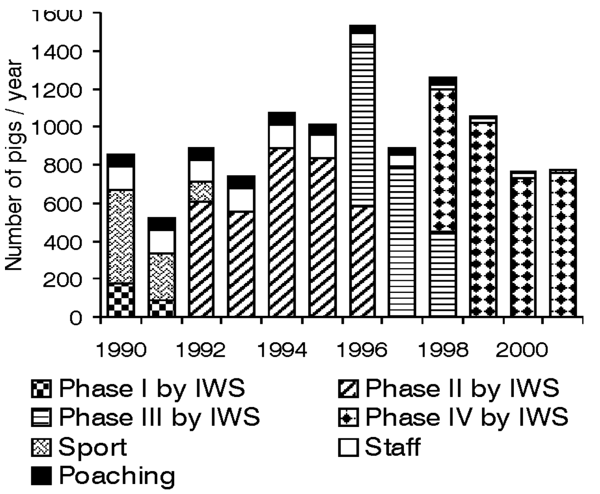

Fig. 5 Number of pigs taken annually on Santa Catalina Island between 1990 and September 2001 by different sources (n = 11,855).

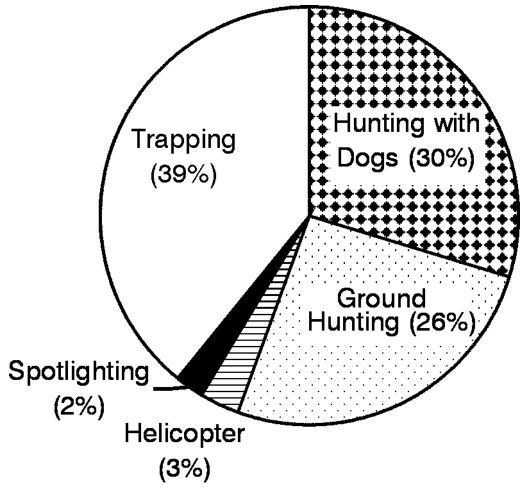

Fig. 7 Percentage of pigs removed by different methods between 1990 and September 2001 on Santa Catalina Island (n = 9123). This includes all kills by IWS as well as a few records by other island residents. The remaining 2732 animals were likely taken by ground hunting by sport hunters, staff or poachers.

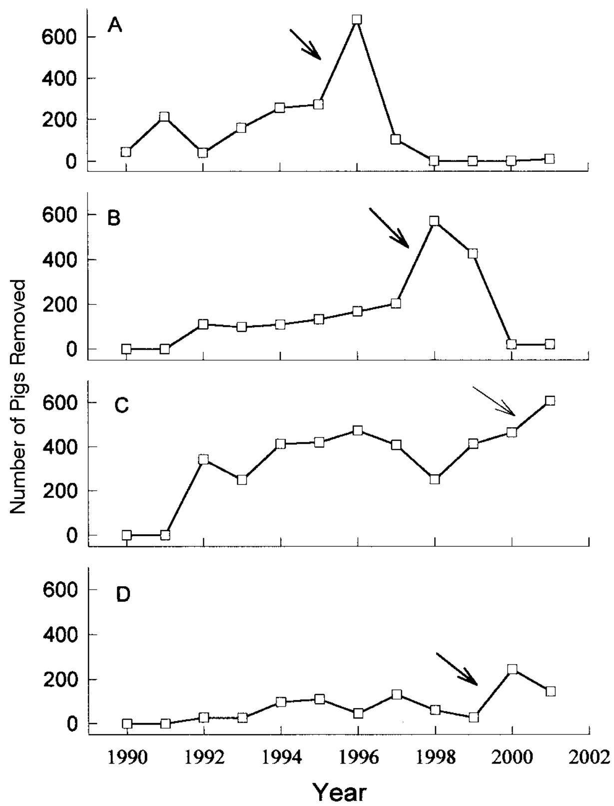

Fig. 6 Number of pigs annually taken from each zone on Santa Catalina Island between 1990 and September 2001; (A) Zone 1: total = 1774, (B) Zone 2: total = 1852, (C) Zone 3: total = 4043, and (D) Zone 4: total = 914. Arrows indicate when objective changed from control to eradication. Accurate locations for 3272 pigs removed before fence construction could not be determined and are not shown.

RESULTS

Eradication techniques

At least 11,855 pigs were killed from 1990 to September 2001. The majority of these (9076) were removed by IWS staff, while the rest were taken by sport hunters, staff hunting, and poaching (Fig. 4). The estimated figures for both staff hunting and poaching probably under-represent the actual take (B. Boyd, D. Gardner, H. Saldaña pers. comm.). Numbers of animals taken during each phase of the programme are shown in Fig. 5. Numbers of animals taken from each zone are shown in Fig. 6.

Removal techniques had differing rates of success and were used in different proportions depending on the season (Fig. 7). Trapping was conducted primarily June through October. Using data collected for the 1998 and 1999 seasons in Zone 2, trapping success (No. traps capturing pigs/ total No. traps set (n = 606)), ranged from 1% to 90% with a mean of 43%. Although pig densities were much higher in 1998, as indicated by the fact that less than 20 pigs were estimated to remain in Zone 2 by late 1999, the seasonal difference in the means were not statistically significant (ANOVA; $p > 0.1$). However, in 1998, 65% of the nights with trapping had >40% success while only 38% of the 1999 nights had >40% success. Numbers of pigs caught in an individual trap at one time ranged from one to 22 during the course of the entire programme, with more than one pig being caught 45% of the time during the two seasons with recorded data (Fig. 8). There were no significant differences in frequency of multiple captures between the two seasons (ANOVA; $p > 0.1$). Dogs were used almost exclusively during the cooler winter and spring months of November through April. Ground hunting occurred year round on an opportunistic basis as animals were

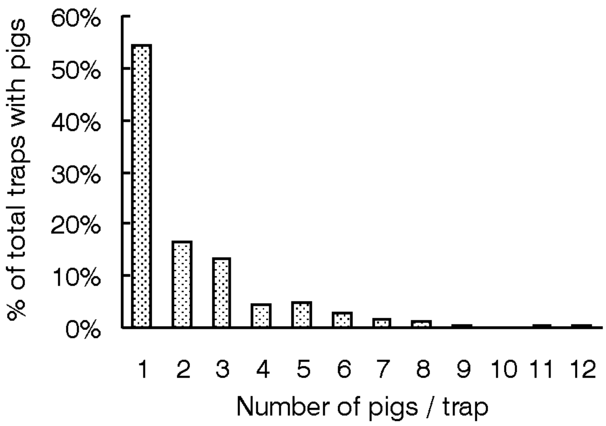

Fig. 8 Percentage of traps with single or multiple captures during the 1998 and 1999 trapping seasons in Zone 2, Santa Catalina Island (n = 268).

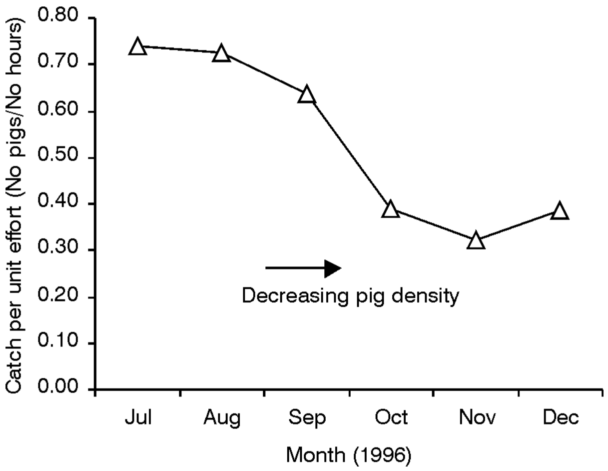

Fig. 9 Monthly catch per unit effort for July-December 1996 during initial Zone 1, Santa Catalina Island pig eradication efforts (n = 655 pigs, 1153 hours). Over 85% of all pigs eventually removed from the zone were taken during this period. Several removal techniques were used.

encountered. Helicopters were used infrequently, usually during the winter months when on the island for goat control efforts.

Catch per unit effort (CPUE) in Zone 1 decreased as the pig population went from high density to low density (Fig. 9). Monthly CPUE for several combined zones over a three year period also shows a steady decrease ranging from a high of 1.05 pigs/hour to a low of 0.03 pigs/hour. Total time efforts expended in Phases I - IV were tracked by different people in varying ways over the course of the programme. Table 2 summarises estimated totals for the programme period. Total annual effort in relation to annual take has not been calculated.

Ensuring the integrity of the fences has been adequately met using 15 volunteers who each assumed responsibility for a section of fence and checked it at least once a month.

Table 2 Total estimated effort expended on Santa Catalina Island from 1990 to September 2001 to remove feral pigs. Only Conservancy and IWS staff time shown. Time spent hunting by sport hunters, Conservancy staff or poachers is not included.

Activity	No. of person-hours expended
IWS field work	64,580
IWS administration	5265
Fence building	
- 1990 Isthmus fence upgrade	250
- 1999 fence construction	12,000
- fence monitoring (1998-Sep 2001)	1260
- fence repair (1998-Sep 2001)	775
Conservancy administration (planning and management)	6380
Education/Public relations	515
Conservancy general support: includes road repair, vehicle maintenance, material transport, fence volunteer management	5475
Total hours	96,500

Although fences suffered only minor damage due to natural or pig-related causes, vandalism of the fences occurred on a number of occasions, particularly just after their construction. Gates were left open and sections of fence, up to 15 m, were cut and removed. Fortunately, the combination of buffer zones along the fences coupled with regular monitoring and immediate repair of vandalised sections prevented any significant movement of pigs between zones.

Current status

Since this is an ongoing programme with an expected completion date no earlier than 2004, there are still pigs remaining on Catalina. Estimated pig numbers as of 1 October 2001 are no more than two in Zone 1, 15 in Zone 2, 200-250 in Zone 3, and 30 in Zone 4.

Monitoring and evaluation

Methods established for monitoring and detection have so far proven effective. Continual scouting of suspected refuges or known areas of high density generally produced the last few pigs. Regular running of dogs through an area during winter months is also effective in finding pigs when numbers are extremely low. Regular systematic hunting in Zone 1 concluded in late 1998 because no animals were known to remain and no additional sign could be found. For the next 18 months, periodic monitoring continued even in the absence of any pig sign. In late summer 2000, pig sign was observed which, while not fresh, was still less than a year old. Scouting efforts were intensified and in November 2000 a solitary sow weighing 80 kg was removed. The sow was not pregnant and did not appear to have given birth in the previous season.

Table 3 Estimated costs (USD) for feral pig removal efforts on Santa Catalina Island from 1990-September 2001. * = combined personnel and operating costs, ** = estimated cost, * = value of volunteer labour. Costs reflect only animals removed by IWS. Dollar figures not adjusted for inflation.**

Activity	Cost/Value in U.S dollars
IWS Field staff costs	
- 1990*	10,000
- 1991*	15,000
- 1992*	66,000
- 1993*	72,200
- 1994	51,958
- 1995	47,760
- 1996	76,756
- 1997	91,597
- 1998	113,753
- 1999	151,161
- 2000	162,996
- 2001 (Jan.-Sep.)	159,863
Control methods	
- helicopter	14,725
- traps - construction	60,283
- traps - bait	17,937
- dogs (maintenance costs 1990- Sep. 2001)	138,982
Fence building	
- Isthmus fence upgrade and repair**	10,000
- 1999 fence construction	
- - contract labour	520,000
- - materials	325,000
- - Conservancy preparatory fence work	18,326
- - Conservancy administration**	10,000
- - volunteer help (81.5 hours)***	1208
- fence monitoring (1252 hours)***	18,900
- fence repair (1999 - Sep 2001)**	38,238
Other	
- kennels (site preparation, construction)	34,412
- vehicles	
- - Four-wheel-drive trucks	169,433
- - All terrain vehicles (ATV)	15,900
- value of housing and office (1-3 houses/yr)	182,690
- general supplies and operating	71,863
- overhead (IWS contract)	283,926
- Conservancy administration**	191,000
- education/public relations (time, materials)**	13,000
- Conservancy general support**	99,093
- gasoline	57,194
- utilities	25,566
- miscellaneous volunteer (38 hours)***	570
Total	3,175,517

Table 4 Age class, sex, and average weight from feral pigs killed on Santa Catalina Island from 1990-September 2001. Sample size (n) for age class = 5777 pigs, n for average weight = 5697, n for sex ratio = 5509. Piglets were defined as <2 months, juveniles were 2-7 months, and adults were >7 months old.

Age class	% of population	Average wt (kg)	Male	Female	Sex Ratio (M:F)
Piglet	18%	0.5	496	482	1:0.97
Juvenile	38%	15.7	1105	1015	1:0.92
Adult	44%	32.1	1244	1167	1:0.94
Totals	100%	20.8	2845	2664	1:0.94

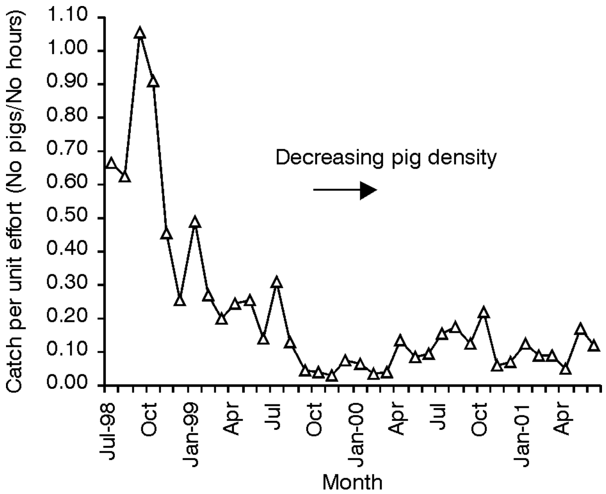

Fig. 10 Monthly catch per unit effort (CPUE) by IWS staff on Santa Catalina Island from July 1998 to June 2001 (n = 3029 pigs, 20 130 hours). Although overall pig densities decreased throughout the time period, making long-term CPUE comparisons is difficult because removal efforts annually shifted locations and seasonally changed removal methods.

Costs

There is a high cost associated with operating a removal programme of this magnitude and length. Known and estimated costs are shown in Table 3 and reflect only costs associated with pig removal efforts conducted by IWS. Current cost per animal averages out to approximately USD350/pig. If the programme stays on schedule, an estimated additional USD825,000 will be needed, bringing a final estimate for cost/pig to USD425.

Population parameters

Age, average weight and sex were noted for killed pigs whenever possible (Table 4). Data were not collected from pigs shot from helicopters nor those removed through sport hunting and poaching. Of the pigs sampled, 87% were estimated to be in average to good condition (Fig. 11; Table 5).

DISCUSSION

Removal efforts

We believe one key to the success of the removal programme to date has been the application of several different techniques. As Fig. 10 shows, trapping is successful and efficient in removing large numbers of pigs during the dry months when food resources are low; the lower the natural food supply, the better the trapping success. We recommend this technique as the starting point for managers facing a high number of pigs in a habitat with seasonal fluctuations in the food supply. For those countries or situations where poisoning is legal and feasible, poisoning could be even more effective than trapping as the initial removal method, as the poison could easily be placed in the bait and no traps would be needed. However, poisoning raises new issues, such as legality, effects on non-target species, and the humaneness of the technique. Since field poisoning of pigs is not a legal option in the United States, we did not consider it.

Dogs are less effective during the dry period and run the risk of becoming severely affected or incapacitated by barbed seeds or foxtails (the flowering heads of a number of grasses) working their way into the eyes, ears, foot pads or other open parts of the dog. During winter months, when pigs are less likely to enter traps if adequate natural food sources are available, dogs are more effective. The scent of the pig is held better on damp soil and vegetation allowing easier tracking. Hunting with dogs is useful not only during winter months when traps are less effective, but also when pig densities are low. It requires knowledgeable dog handlers and at least some "lead" dogs. An experienced "lead" dog with a sensitive nose will relentlessly search for pigs and engage the animal when located. Non "lead" dogs are still beneficial as they will follow and assist in holding the pig at bay until the hunter arrives.

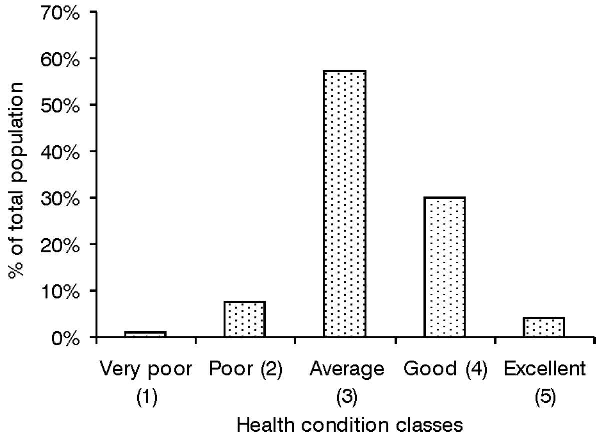

Fig. 11 Percentage of pigs on Santa Catalina Island in different health condition classes between 1994 and September 2001 (n= 5207). Class ratings based on subjective scale of very poor (1) to excellent (5) (See Table 5).

Each of the other methods used (spotlighting, ground hunting, aerial hunting, night vision hunting) provided a contribution to the programme, but each had constraints on their effectiveness depending on time of year, density of pigs, and/or physiographic features in the areas they were being employed.

Spotlighting is an effective technique at all pig densities providing a good road system is present, and tall vegetation near the roads does not obscure the shooter's line of vision. In habitats with low-growing ground cover, pigs can be shot at more than 200 m from the roads using this method. Ground hunting, which includes opportunistic shooting of pigs when conducting other activities, is likewise effective at all pig densities. However, due to safety concerns, shots could often not be taken when pigs were encountered. Aerial hunting from a helicopter can be effective in areas lacking dense woody vegetation during

Table 5 Criteria for assigning condition class for feral pigs killed on Santa Catalina Island. Criteria and averages based on Timm *et al*. (1994). Assessing the condition of these animals is subjective, particularly if animals exhibited traits in more than one condition.

Condition Class	Parasite Burden	Coat Condition	Fat Reserves	Average size for age and sex	Other physical and visual signs
Very poor (1)	very high	thin, sparse	emaciated; no fat reserves	stunted	obvious lesions and medical problems
Poor (2)	high	patchy	thin; visible bony points	<average	often disease processes evident
Average (3)	present	normal	normal muscling; lean with some fat reserves	average	no major medical problems
Good (4)	few	good	well muscled, points felt but not seen	>average	only minor visual signs of any disease
Excellent (5)	none/few	thick	very well muscled; abundant fat reserves; bony points not felt	large	no evidence of any problems

the cooler months of the year when pigs are more likely to be active during daylight hours. FLIR technology was not very successful on Catalina, probably due to heavy vegetation. Although effective, aerial hunting repeatedly caused more public concern than any other method.

Since the data collected and recorded was not designed to allow for direct comparisons of the catch per unit effort between each method, the results presented should be treated as estimates of the relative value of each technique. Due to multiple captures in a single trap, trapping accounts for large numbers of pigs early in a programme when densities are high and maintains effectiveness even during a lower-density second trapping season**.** Although the CPUE drops when using dogs, managers must remember several important factors: (1) pig densities are generally already low when dogs are used since work in all zones started with summer trapping, (2) use of traps during wet months would likely produce results with an even lower CPUE since most pigs are not interested in bait during the wet months, and (3) some pigs are "trap shy" and may not enter regardless of the bait or design used. We found the use of dogs after initial trapping to be a critical component of our success and generally increased the CPUE for a given area when first used.

Programme commitment

Even more important than any particular removal technique is the need to have organisational consensus on programme goals and full commitment and support to ensure the project is completed. It was not until the programme goal changed from control efforts to eradication that significant declines in pig numbers could be noted for any zone. Reviewing the numbers in Fig. 6, annual take remained relatively constant by zone until the goal shifted to eradication. Efforts in Phase I were effective in reducing pig numbers to less than 50 pigs in Zone 1. However, the 10 month break that occurred negated all Phase I efforts, and it was not until Phase III in 1996 that a sustained decline was again observed in this zone. Once the attempt to eradicate a pig population in an area has started, it is important to maintain or even increase efforts in order to reach the goal of zero animals. The alternative is to waste financial resources, jeopardise the natural resources we are seeking to protect, kill many animals with no lasting benefit to the resource, and possibly lose the momentum required to make the programme a success.

All programmes of this magnitude and duration will experience delays or setbacks during implementation. We experienced delays due to: (1) changing objectives on the part of some board members, (2) occasional financial constraints, (3) reduced access to areas after the 1997 El Niño winter produced near 100 cm of rain, (4) vandalism to traps and fences, and (5) occasional high turnover in staff. Without long-term commitment to the project, these types of natural and anthropogenic factors could have serious implications to the success and continuation of the programme.

Monitoring and search effort

The need for continual monitoring long after all animals are suspected to be removed is vital to success. Shutting down a large programme only to find a need to gear back up after several years could prove to be impossible if the commitment has waned or staff has moved on in the intervening years. A programme should be maintained until it is certain all pigs have been eradicated. In Zone 1, for two years we thought no animals remained, yet a 80 kg sow then appeared. Such a large animal escaping detection for an extended period of time indicates the need for monitoring to continue for a significant period after the last known sighting of any animal or sign.

Public relations and education

The need to have public relations and educational components of a control programme cannot be overemphasised. We found it is easier to be forthcoming with all aspects of a programme prior to implementation, rather than trying to defend past policies and practices. By implementing the programme in phases we had the opportunity to experience both situations in this programme. Phases I, II, and III were not actively brought to the public's attention whereas most of Phase IV had extensive prior exposure. There will always be a segment of the population who take issue with the need for such projects. We had unknown persons either capturing piglets on one part of the island or bringing them from the mainland and then releasing them in controlled areas. As long as animals remain anywhere on the island, this remains a possibility and dictates that both education and monitoring efforts need to continue long after the end of intensive field efforts. We think working with a well-researched and articulated set of goals and objectives with an informed public is better than allowing rumours and misinformation to shape the public's awareness.

Management implications

The methods used to eradicate feral pigs from Catalina are proving successful and we expect to complete the project in the next few years. The methods and results have application for eradication programmes elsewhere, although site-specific conditions will obviously play a large role in their timing, duration, and intensity. If eradication is the goal, a committed, high-intensity, well-funded effort is essential, and in the long run will be more labour efficient and cost-effective than long-term or sporadic control. We have found that all the results from even relatively-intense control efforts can be quickly lost if continual and increased pressure is not applied to achieving complete removal. An intensive, shorter programme will also lower the total number of animals needing removal. Managers who do not have the option of complete removal must seriously evaluate methods, costs, and expected results to find a programme that can be efficiently and humanely continued year in and year out.

If the Conservancy had to do this programme over, three key changes would be made. First, the programme would have been clearly defined as an eradication effort from the start. This would have reduced the overall length of the programme, reduced the amount of time subject to public relations issues, and reduced the cost of the programme to perhaps USD375/pig rather than the projected USD425/pig. Second, data collection protocols would be modified to gather more accurate figures for total costs, and total time expended as well, determining capture per unit efforts and percentage success for different methods and differing pig densities. Third, a public education programme would have been implemented prior to any control efforts and would have been continued for the duration of the entire project.

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Eradication of potentially invasive plants with limited distributions in the Galapagos Islands

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Abstract A cooperative project between the Charles Darwin Foundation and the Galapagos National Park Service has been initiated to attempt to eradicate several populations of potentially-invasive plant species from the Galapagos Islands. More than 600 introduced plant species have been recorded in Galapagos, of which many are already serious invaders. Among the cultivated and recently naturalised species, many are potentially invasive, but still have limited distributions and can be eradicated. This paper discusses attempts at plant eradication in the Galapagos using three examples with differing degrees of invasiveness. A priority list of species to be eradicated is being compiled by means of a risk assessment system based on a database, literature, local knowledge, ongoing surveys and information from elsewhere. The target plants are then mapped. If an effective control treatment is known for a particular species, the field team performs the eradication work. If not, trials are conducted to determine the best technique. Once removal has been carried out, locations are monitored at appropriate intervals until the plant has not been recorded for at least three years. *Pueraria phaseoloides*, a known invasive vine, was recently introduced at a single site (0.04 ha) and has not been seen again since it was last treated in 1997. *Rubus glaucus*, a potentially-invasive scrambler, was introduced more than 25 years ago and is sparsely distributed over about 5 ha. The timber tree *Citharexylum gentryi* was introduced in 1950 but was only recorded by scientists in 1999. It has many invasive characteristics, has mature reproductive stands and is distributed over about 171 ha. All known reproductive individuals of both *R. glaucus* and *C. gentryi* have now been removed and monitoring continues.

Resumen Un proyecto cooperativo entre la Fundación Charles Darwin y el Parque Nacional Galápagos se ha inicializado para intentar erradicar algunas poblaciones de especies invasivas de Galápagos. Más de 600 especies de plantas introducidas han sido registradas en Galápagos, de las cuales muchas son catalagodas como altamente invasivas. Entre las especies cultivadas y recientemente naturalizadas muchas son potencialmente invasivas, pero tienen distribuciones limitadas y aun pueden ser erradicadas. El propósito de este documento es discutir los esfuerzos de erradicación de plantas usando tres ejemplos con diferente grado de invasividad. Se selecciona una lista de especies prioritarias a ser erradicadas a través de un conjunto de criterios que juzgan el riesgo de invasividad de cada especie basado en; la información de la base de datos, bibliografía, conocimiento local, monitoreo e información de otros sitios. Las plantas seleccionadas son posteriormente mapeadas. Si el tratamiento de control efectivo es conocido para una especie particular, el personal de campo realiza el trabajo de erradicación. Si no, experimentos son conducidos para determinar la mejor técnica. Una vez que el trabajo inicial ha sido llevado a cabo, las poblaciones son monitoreadas cada tiempo apropriado hasta que ningun individuo sea observado por tres años consecutivos. *Pueraria phaseoloides* una trepadora conocida como invasiva fue recientemente introducida en un solo sitio y no ha sido observada otra vez desde que fue tratada en 1997. *Rubus glaucus*, una especie potencialmente invasiva, fue introducida hace más de 25 años y está distribuida de forma esparcida en algo menos de 5 ha. El árbol maderable *Citharexylum gentryi* fue introducido en 1950, pero fue registrado por primera vez por científicos en 1999. Tiene algunas características invasivas, forma grupos de individuos maduros y reproductivos y está dispersado sobre unas 171 ha. Todos los individuos reproductivos de *R. glaucus* y *C. gentryi* localizados fueron removidos y el monitoreo continúa.

Keywords Galapagos; eradication; invasive plant; *Citharexylum gentryi; Pueraria phaseoloides*; *Rubus glaucus.*

INTRODUCTION

One of the most serious threats to the unique flora and fauna of Galapagos is invasion by introduced plants. Over 600 introduced plant species have been recorded in Galapagos (Tye *et al.* 2002) of which 45% may be naturalised (cf. Mauchamp 1997). The areas with the biggest invasive plant problems are principally in the humid highland regions of the inhabited islands. Five of the islands have permanent human communities: Floreana, San Cristobal, Isabela, Santa Cruz and Baltra (which is an arid island with a military base and airport). Even though the populated areas (i.e. not National Park) take up less than 4% of the archipelago, they have disproportionately affected the restricted and vulnerable highland areas.

Most potentially-invasive plants have been introduced deliberately; therefore, the agricultural zones act as a source of spread to adjacent protected areas. Most invasions commence in the urban and agricultural zones with plants principally dispersing into the National Park along paths and roads (Schofield 1973; Jaramillo 1999). For example, species such as *Urochloa brizantha*, *Abrus precatorius*, *Dalechampia scandens*, and *Leucaena leucocephala* have dispersed from the agricultural zone and are starting to invade the arid and semi-arid areas of Santa Cruz. The

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most graphic example of dispersal is *Cinchona pubescens*, of which a few trees were introduced in 1946 (Jäger 1999), and are responsible for the invasion of more than 11,000 ha of the humid highlands in Santa Cruz.

Quantitative studies on the impacts of widespread invasive species have shown that the distribution and abundance of native species have been seriously changed. Jäger (1999) showed that *Cinchona pubescens* severely affected the native vegetation in both the *Miconia robinsoniana* and the fern-sedge vegetation zones. The invasion of the tree *C. pubescens* also threatens populations of rare endemic herbaceous species with restricted distribution such as *Pernettya howellii* and *Acalypha wigginsii*. Other invasive species such as *Psidium guajava*, *Lantana camara*, *Syzygium jambos*, *Pennisetum purpureum* and *Rubus niveus* are widely dispersed in four of the populated islands (Lawesson and Ortiz 1994). These are altering native ecosystems and causing some economic losses to the agricultural sector. The principal invasives are trees, scramblers, climbers and grasses (Tye *et al.* 2002). Trees, in particular, are a threat as the native vegetation rarely exceeds 10 m in height and the humid highlands are covered with low scrub and herbaceous vegetation. Therefore, to protect the native ecosystem it is imperative to eradicate or contain potentially-invasive species with limited distribution before they become widespread. Attempting to eradicate populations that are restricted in distribution is much more cost-effective than long-term control and has a high probability of success.

In the year 2000 an inventory of the agricultural zone of the island of Santa Cruz was completed, taking the number of introduced species in Galapagos to over 600. One hundred previously-unrecorded species were found, of which some are already known to be invasive in other parts of the world. An example of differences in establishment time is *Psidium guajava*, one of the most invasive plants in Galapagos. This species was introduced to San Cristobal Island in about 1869 and was restricted to small plantations. It did not become invasive until the 1950s (80 years later). However, in Santa Cruz it was introduced in 1930 and became invasive within only 40 years. Another example is *Lantana camara*, which was introduced to Floreana in 1938 as an ornamental but did not become invasive until 1970 (Eckhardt 1972).

In order to select which species to include in the eradication programme, we are developing a system to prioritise our eradication activities (Tye 2001; Tye *et al.* 2002) based on distribution, plant biology, potential invasiveness (both in Galapagos and elsewhere), availability of treatment methods, and ease of treatment. If potential invasives are treated during the establishment or 'lag phase' there is a much higher probability of eradication. Also, species that are not utilised by the local community are selected for eradication. This guarantees the support of the community and reduces the risk of re-introduction.

A programme to eradicate several species has been initiated in Santa Cruz Island and is to be expanded to other islands. The programme commenced in Santa Cruz because most resources are there and it is the island with the most complete invasive database. Target species include the trees *Citharexylum gentryi* and *Leucaena leucocephala*, several scramblers in the genus *Rubus*, and climbers such as *Dalechampia scandens* and *Pueraria phaseoloides*. *Rubus* spp. in particular are known as invasives worldwide. In Galapagos the most invasive is *Rubus niveus*, which is present on three islands and is distributed over more than 10,000 ha, but has a limited distribution on Isabela Island. *Rubus adenotrichos*, *R. glaucus* and *R. megalococcus* all still have limited distributions and are currently being targeted by the eradication programme.

This paper outlines the methodology and success in eradication of three species with differing distributions but confined to Santa Cruz: the climber *Pueraria phaseoloides* (Fabaceae), the scrambler *Rubus glaucus* (Rosaceae) and the tree *Citharexylum gentryi* (Verbenaceae). These species were selected because they were considered potentially invasive or showing signs of becoming so, and the probability of eradication success is high.

Pueraria phaseoloides: This species (tropical kudzu) is native to Southeast Asia and is used as ground cover to fix nitrogen and as a forage plant. The USDA (2001) has listed it as a noxious weed because of its invasive potential. It is related to the highly-invasive *Pueraria lobata* ohwi (kudzu) which is one of the most serious pests in southeast U.S.A. *Pueraria phaseoloides* was introduced by one farmer in 1996 and was only found in one location (to our knowledge). It is located in the agricultural zone (450 m altitude) in a pasture dominated by the introduced pasture grass *Urochloa brizantha*.

Rubus glaucus: This species (mountain blackberry) is native to the Andes in northern South America and was introduced into the Galapagos sometime before 1974. It is the only species of this genus that is commonly cultivated in Ecuador and used commercially for its edible fruits (Romoleroux 1996). It is only naturalised within the National Park in the north-west highlands of Santa Cruz, in an area previously used for agriculture but which was incorporated into the National Park in 1974. This species is present in several Pacific islands and is considered by Sherley (2000) as having serious potential as an invasive. *Rubus glaucus* has the potential to spread in both native forests and fern/grasslands. It occurs in well-drained soil from 600-700 m altitude. In Santa Cruz, the representative native species are the tree *Scalesia pedunculata*, the shrub *Tournefortia rufo-sericea*, the herbs *Alternanthera halimifolia* and *Pilea baurii*, and ferns such as *Adiantum henslovianum* and *Blechnum occidentale*.

Citharexylum gentryi: This species (white wood) is a 20 m tall tree native to lowland coastal and Amazonian Ecuador and is common in humid and littoral forests (Jørgensen and León-Yanez 1999). The seeds of *C. gentryi* were introduced accidentally around 1950 in the leaves of a bromeliad that were used as a living fence (D. Uribe pers. comm.). It apparently has medicinal properties and can be used as an anti-inflammatory (D. Uribe pers. comm.). *Citharexylum gentryi* is naturalised in the agricultural lands of Santa Cruz and has a huge potential to spread into the transition zone dry forests of the National Park. This species is considered highly invasive as it is reproductive from a young age, produces many fleshy fruits, has great dispersal ability, and can colonise relatively-undisturbed areas. Two other species of the same genus, *C. spinosum* and *C. caudatum*, are invasive in Pacific islands such as Fiji, French Polynesia, and Hawaii, with *Citharexylum spinosum* mainly invasive in arid habitats below 500 m (Smith 1985).

METHODS

The process of eradicating (localising, treating, and monitoring) the three selected species *Pueraria phaseoloides, Rubus glaucus* and *Citharexylum gentryi* is outlined below. Since little is known about these newly-discovered species in the Galapagos, some background information is presented in the results.

Locating and monitoring target species

A survey of the farms of all sectors of the agricultural zone of Santa Cruz was carried out in 2000 and special attention was paid to the target species. All landholders in this survey were questioned to find out if they had these species or any other unknown invasives.

Pueraria phaseoloides: The plant was identified in 1996 by a Charles Darwin Research Station (CDRS) botanist. Adjacent fields and farms were thoroughly searched and other farmers in the community were questioned as to whether they had sown these species. After the initial treatment in March 1996, the site was revisited initially at two-month intervals, later extended to every six months. The last known plants were sprayed in September 1997 and the site has been visited at yearly intervals since.

Rubus glaucus: This was first identified in 2000 by CDRS botanists. Most infestations consisted of thickets that were between 3 m and 10 m in diameter. An area of about 20 ha was extensively searched around the infestations and along adjacent watercourses. A series of GPS-directed 500 m transects with monitoring stations every 50 m (a total area of 100 ha) was laid out along the altitudinal contours. Monthly visits were made to the site from April 2000 (except September) to January 2001.

Citharexylum gentryi: This tree was first brought to the attention of CDRS in 1999. Two methods were employed to search for untreated trees during post-treatment monitoring. Firstly, a radius of about 100 m around known (treated) trees and areas between patches of trees were searched intensively. Secondly, an area of 1200 ha based around the known infestations and areas that could potentially have other plants was searched. Over a five-week period a series of GPS-directed transects 100 m apart was walked by trained observers looking for adults and seedlings. Monthly visits were made to the site from April 2000 (except September) to January 2001. After elimination of adult individuals (by January 2001), follow up to control seedlings is being done every three months.

Techniques for control of target species

There were two stages to control operations: firstly to remove all seed-producing adults and then seedlings and immature plants. Once no more individuals were found in a given area a system of monitoring was initiated in the local area to look for further individuals. Chemical control methods were used, including for seedling control, since this is faster and cheaper than manual control, and none of the species were growing in highly-sensitive natural vegetation. A limited number of herbicides are available in Ecuador, which prohibits importation of unregistered products, and stricter controls exist for Galapagos. Hence, for this programme we have only used products that are commercially available, effective, and cause the minimal possible environmental damage. Similarly, we try to use application methods that are logistically easy, and t have minimal impact on the surrounding native vegetation.

Pueraria phaseoloides: The 20 x 20 m plot and later regrowth was spot sprayed with a 5% solution of Roundup™ (Monsanto) (41% glyphosate salt) between March 1996 (initial treatment) and September 1997 (follow-ups).

Rubus glaucus: Thickets were sprayed with a 2% solution of Roundup (41% glyphosate salt) until leaves were wet. Germinating seedlings were sprayed with a 1.5% solution of Roundup. Treatment commenced in April 2000 and is continuing.

Citharexylum gentryi: All individuals were treated with the product Combo™ (Dow) a two-part mix (267 ml: 6 g) of picloram salt (24% v/v) and metsulphuron methyl (66% w/w). The product was dissolved in freshwater. Seedlings were sprayed with a 1% solution of Combo. Individuals between 1 cm and 30 cm in diameter were cut down using a machete or chainsaw as close to the ground as possible and the cut area (particularly the bark) was painted immediately with a 5% solution of Combo. Individuals greater than 30 cm in diameter were similarly treated with a 10 % solution of Combo. Regrowth from trunks was sprayed with a 10% solution of Combo. Treatment commenced in April 2000 and will continue until no seedlings have reappeared for at least three years.

RESULTS

In summary, *P. phaseoloides* has not been seen since 1997 (after three years it was declared eradicated) and all known seed-producing individuals have been removed of *R. glaucus* and *C. gentry*i (Table 1).

Table 1 Eradication effort (includes control and monitoring) for three species of potentially invasive species in Galapagos.

Species	Habit	Infestation area (ha)	Eradication status to date	Effort (person-hours to April 2001)
Pueraria phaseoloides	Climber	0.04	Eradicated	120
Rubus glaucus	Scrambler	5	No reproductive individuals	565
Citharexylum gentryi	Tree	171	No reproductive individuals	2870

Pueraria phaseoloides: All plants in the 0.04 ha patch have been destroyed and the last individual was seen in September 1997. Foliar application of 5% Roundup resulted in 100% mortality of adult plants. Plants were observed flowering but never produced any mature fruits. To date the only known dispersal mechanism of this species in Galapagos is humans. A total of 120 person-hours was spent in community consultation, treatment, and monitoring of this plant.

Rubus glaucus: Was reported for the first time in February 2000, although local people had known of its existence for some time (probably planted in the 1960s or 1970s for fruit production). An infestation of approximately 5 ha was found. About 100 ha of the surrounding area were searched using a grid of equidistant points, but no further plants were found. Plants are generally located along watercourses. It was observed to produce fruits during September and these were removed. Several treatments and monitoring visits were made between April and November of 2000. All adult plants were eliminated and no new adults have been found since. However, seedlings have been subsequently found and treated. The foliar application of 2% Roundup resulted in high mortality of adult plants but also resulted in the death of much surrounding vegetation. However, good regeneration of *Scalesia pedunculata* was recorded on the bare ground. A total of 490 person-hours has been spent so far on the treatment and monitoring of this plant.

Citharexylum gentryi: Is dispersed over an area of 171 ha in the agricultural zone of Santa Cruz, and a single tree has been found in the National Park. Over 1200 ha have been searched for this species. It is located between 100 and 400 m in altitude in two zones in the sector of Bellavista (the southern side of the agricultural zone) and in Camote (the south-eastern side of the agricultural zone). It occurs generally in mixed forests of introduced trees such as *Cedrela odorata* and *Psidium guajava* but is sometimes found in remnant native forest dominated by *Psidium galapageium.* Abundant black, berry-like drupes, with a single seed up to 6 mm in diameter, are produced for most of the year. In Galapagos, adult trees (i.e. >6 cm in diameter and >3 m tall, producing fruit) can reach a diameter of 1.2 m and a height of 22 m. The mean diameter and height of the treated trees was 19 cm and 7.5 m. Around where the trees were originally introduced the density of adult trees was about 14/100 m^2 whereas away from the founder patch the density was about 5/100 m^2 and the majority of trees were less than 15 cm in diameter.

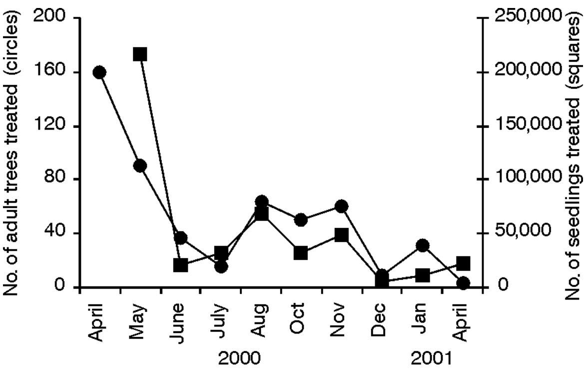

Fig. 1 Total numbers of adult trees (circles) and seedlings (squares) of *Citharexylum gentryi* treated.

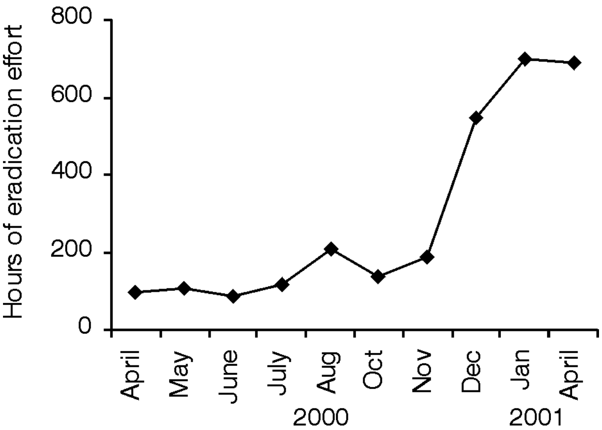

Fig. 2 Person-hours of eradication effort invested in control and monitoring of *Citharexylum gentryi*

Between April 2000 and April 2001 all known seed-producing trees were felled and treated. Figure 1 shows the progressive reduction of the adult individuals and seedlings. A total of 570 adult trees were treated. Once this canopy was removed there was a large flush of seedlings. The number of seedlings treated (approximately 450,000) decreased with time, although monthly effort on this task was kept the same until the last three surveys, when it was increased to ensure that further seedlings at low density would not escape detection (Fig. 2). In some areas of dense infestation, maize was sown to reduce seedling germination. The cut stump method using 5% Combo was effective for smaller trees but those greater than 30 cm diameter often resprouted and needed further treatment (using 10% Combo).

DISCUSSION

This programme demonstrates that eradication of potentially-invasive plants is possible in Galapagos. So far, with relatively few resources, we have managed to eradicate one species and remove all seed-producing adults of two other species. There are some obvious differences between the three examples discussed here, including the establishment time and size of infestation. *Pueraria phaseoloides* was introduced recently, only cultivated in a small area, and was found by botanists almost immediately. It had no chance to reproduce and become naturalised. Assuming that it has not been cultivated elsewhere in the islands, and given that it has not been seen for more than three years in a site that is very easy to monitor, it is safe to pronounce it eradicated. Conversely, *R. glaucus* and *C. gentryi* are both naturalised, have been present for 25 and 50 years, and have distributions of at least 5 ha and 171 ha respectively. *Rubus glaucus* is still in the establishment phase whereas *C. gentryi* is in the expansion phase. Both have produced reproductive offspring and the presence of seedlings suggests that a seedbank has formed. Although it is not known how long the seedbanks of these species persist, a study of *Rubus niveus* in Galapagos showed that after one year in the soil at least 25% of the seeds were still viable (O. Landázuri pers. comm.). *Citharexylum gentryi* has a sizeable hard seed that can remain dormant for at least six months (M. Soria pers. obs.). Hence, before these species can be eradicated the seedbank must be exhausted. This is a difficult task and preventing further seed input requires careful and repeated monitoring for many years. Fortunately, it takes C. *gentryi* from 3-6 years to reach reproductive age (it is far easier to prevent seed production in a slow-growing species that reproduces only after several years than it is for an annual), and few seeds are produced by *R. glaucus* (which can probably reproduce after 12 months), so the effect of missing a few small individuals during monthly monitoring is not great.

The other important factor when considering eradication, is capacity for dispersal. In Galapagos, wind-dispersed species are among the most difficult to control, since there are few bird species that are efficient seed dispersers. Both *R. glaucus* and *C. gentryi* to date have little dispersal potential and are mainly dispersed short distances by gravity. *Rubus glaucus* has a fleshy drupe which is evolved for animal dispersal, but few potential agents exist. The Galapagos flycatcher has been observed to disperse *R. niveus* (A. M. Guerrero pers. comm.) and may also disperse *R. glaucus*. *Citharexylum gentryi* has a berry-like drupe but no animal dispersal has been observed. Dispersal by water seems to have caused the occurrence of these species along streams, which makes them easier to locate.

Success of eradication also depends on accessibility. *C. gentryi* is found in farmland and is easier to locate compared to *R. glaucus* which is found in dense forest on rough terrain within the National Park, which increases the chance of escaping discovery. The *R. glaucus* site is nearly 10 km from the nearest road and all equipment must be carried to it. Both *R. glaucus* and *C. gentryi* will require at least three more years of monitoring and treatment. However, considering their potential invasiveness and the vulnerability of the community they are invading, their eradication must be considered top priority.

In the next six years we intend to expand this programme and attempt to eradicate 30 species of potentially-invasive plants archipelago-wide. Before this can be initiated, a complete introduced species database is required for the four main populated islands. Presently, good data exist for Santa Cruz and Floreana only. Without full information we cannot declare a species eradicated with any confidence. Also sufficient and systematic monitoring is required, to ensure that all individuals of the target species have been discovered. Effective control methods (principally chemical control) need to be developed for many of the lesser-known species as little information is available worldwide on their treatment. A protocol to evaluate eradication success needs to be refined. Another problem is getting community support to work with plants that are useful or are not obviously weeds today. One landholder could not understand how the beautiful little water plant *Eichhornia crassipes* (Martius) Solms could ever be a problem. If *Urochloa brizantha* were ever to be eradicated it would be very difficult to prevent its re-introduction because it is highly regarded for pasture. Therefore, one strong component of the eradication programme is a long-term education campaign explaining the threats of these plants, the production of a list of permitted species, and a quarantine system that prevents further introductions. Although difficult and requiring long-term commitment, this project may actually reduce the number of introduced plant species and will save millions of dollars in future management.

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Island conservation in north-west Mexico: a conservation model integrating research, education and exotic mammal eradication

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Dedication This paper is dedicated to Jesús Ramírez, whose work marked the beginning of island conservation in Mexico.

Abstract The 250+ islands of north-west Mexico support 50 taxa (species and subspecies) of breeding seabirds and over 180 taxa of endemic terrestrial vertebrates. Isolation and aridity have historically protected these islands from many human perturbations and consequently their biotas are relatively intact. However, invasive alien mammals have been introduced to at least 44 islands and are responsible for the ecological extinction of 22 endemic vertebrate species and subspecies, and the local extinction of one or more seabird taxa on 10 islands. The Island Conservation and Ecology Group, the National Autonomous University of Mexico, Center for Biological Investigations, and National Protected Areas Department collaborated with local people and other NGOs to remove one or more introduced mammals from 23 islands and will soon complete eradication on one more. This work has protected habitat for 27 seabird taxa, seven of which are endemic to the region, and 38 endemic taxa of terrestrial vertebrates. Our regional, science-based, collaborative approach to island conservation has eradicated invasive alien mammals from most islands under 40 km^2 in this biologically important region. We are building on this experience to conduct eradications on larger, more-difficult islands.

Keywords endemic species; extinction; extirpation; invasive alien; Baja California; Gulf of California; introduced species.

La conservación de islas en el noroeste de México: un modelo de conservación que integra la investigación, educación y erradicación de mamíferos exóticos

Dedicatoria: Este artículo está dedicado a Jesús Ramírez, cuyo trabajo marcó el inicio de la conservación de islas en México.

Resumen Las más de 250 islas del noroeste de México contienen 50 taxones (especies y subespecies) de aves marinas en reproducción y más de 180 especies y subespecies de vertebrados terrestres endémicos. A lo largo de la historia el aislamiento y la aridez han protegido a estas islas de las perturbaciones y consecuentemente la biota se ha mantenido relativamente intacta. Sin embargo, mamíferos ajenos a islas han sido introducidos a por lo menos 44 islas y son responsables de la extinción ecológica de 22 especies y subespecies de vertebrados endémicos y de la extinción regional en diez islas de uno o más taxones de aves marinas. El Grupo de Ecología y Conservación de Islas, la Universidad Nacional Autónoma de México, el Centro de Investigaciones Biológicas y el Departamento de Áreas Nacionales Protegidas han colaborado con la gente de la región y con otras agencias no gubernamentales para remover una o más especies de mamíferos exóticos de 23 islas y muy pronto de otra más. Este trabajo ha protegido el hábitat de 27 especies y subespecies de aves marinas, siete de las cuales son endémicas de la región, y de 38 especies y subespecies endémicas de vertebrados terrestres. Nuestro enfoque hacia la conservación de islas es regional y con base en investigación científica y la colaboración, y ha removido los mamíferos exóticos de la mayoría de las islas menores a 40 km^2 en esta importante región biológica. Esta experiencia nos permitirá llevar a cabo erradicaciones en islas más grandes y difíciles.

INTRODUCTION

Many island ecosystems lack native terrestrial mammals (Carlquist 1974). This unique evolutionary history makes island ecosystems particularly vulnerable to the impacts of invasive alien mammals because: (1) native island species generally have poor behavioural, physical, and life history defences against mammalian herbivory and predation (Mooney and Drake 1986; Stone *et al.* 1994; Bowen and Van Vuren 1997), and (2) island ecosystems typically

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lack native predators that can regulate invading mammal populations. Consequently, the introduction of alien mammals to island ecosystems is one of the greatest causes of recorded global extinctions (Elton 1958; King 1985; Atkinson 1989; Diamond 1989; Groombridge *et al.* 1992).

The more than 250 islands in north-west Mexico are known for their high biodiversity, endemism, important seabird colonies, and relatively-low levels of human disturbance (Case and Cody 1983; Everett and Anderson 1991; Ceballos *et al*. 1998; Alvarez-Castaneda and Patton 1999; Grismer 1999; Donlan *et al*. 2000). Historically, these islands were protected from most direct human perturbations by aridity, isolation, and low human population densities on the adjacent mainland (Tershy *et al.* 1997). Today, most of the islands are government owned and legally protected from many forms of land conversion (Carabias-Lillo *et al.* 2000). However, alien mammals were introduced to many of these islands starting in the late 1800s and early 1900s (Jehl and Parkes 1982; Jehl and Everett 1985; Brattstrom 1990; Martinez-Gomez and Curry 1996; McChesney and Tershy 1998). Alien mammal introductions continue to take place and this threat is exacerbated by dramatic increases in human use of the islands over the last 30 years (Velarde and Anderson 1994; Tershy *et al.* 1999).

To prevent extinctions and protect natural ecological and evolutionary processes, we have been studying and removing invasive alien mammals from islands in north-west Mexico since 1994. In this paper we review the distribution and impacts of introduced mammals on these islands, and summarise our alien mammal eradication projects. We discuss our regional conservation model that integrates applied research, environmental education, and invasive alien mammal eradication.

METHODS

Approach

We formed a bi-national, non-profit conservation group, the Island Conservation and Ecology Group, to conduct science-driven conservation and applied research on the islands of north-west Mexico. In order to help prioritise our efforts, we developed a conservation biodiversity database. This public database, available via the internet, serves as a central data location, holding referenced data on the distribution and abundance of both native and alien species across multiple taxa, as well endemism levels, island geography data, and human use data (http://www.islandconservation.org, Donlan *et al*. 2000). This database was used to help prioritise islands for alien mammal eradications, based on their biodiversity, the potential impacts of alien mammals, and the political and technical feasibility of the eradication. Once islands were identified for eradication, we implemented local environmental education programmes to gain community support for eradications and worked collaboratively with local and national management agencies on each island (Donlan and Keitt 1999). Our Mexico branch developed the environmental education programmes, which included on-island presentations; school field trips to encourage appreciation of native biodiversity; island conservation education materials such as bumper stickers, videos, and books; and actively involving island residents in aspects of selected projects.

For actual eradications, we trained local biologists to remove introduced rodents from islands and recruited and trained local hunters and trappers for larger alien mammal eradications (i.e., feral cats, goats, and rabbits). Our eradication efforts began on small islands (<3 km^2), that were used by only a limited number of people; hence, both eradication and gaining complete community support were relatively easy and inexpensive. With experience and success on small islands, our hunting/trapping team grew in numbers and experience, and our relationships with agency staff and funders developed. This enabled us to work on progressively larger and more politically-complex islands, and to work on multiple islands simultaneously. In conjunction with the eradication programme, we developed a research programme designed to study the impact and recovery of island ecosystems from exotic mammals (Keitt 1998; Donlan 2000; Keitt *et al*. 2000a, 2000b; Donlan *et al*. 2002; Roemer *et al*. 2002).

Distribution and impacts of alien mammals

To summarise the distribution and impacts of alien mammals on the islands of north-west Mexico we relied on published and unpublished literature including museum specimens, historical records, personal communications from researchers and island residents, and our own field notes. A brief visit to an island was usually sufficient to confirm the presence or absence of larger alien mammals. Also, local island users or other researchers could reliably report on the presence or absence of rabbits, cats, goats, sheep, pigs, or donkeys. Determining the status of introduced rodents on the islands proved more difficult; most island users, including researchers, could not reliably distinguish between native and alien rodents. We visited islands that were suspected of having introduced rodents, and live-trapped (most islands have native rodents) for several nights (2 – 5 nights) to confirm their presence. Not all islands have been surveyed for introduced rodents and a systematic survey would likely result in additional records.

To measure some of the impacts of alien mammals on these island ecosystems, we identified islands where native vertebrate species and subspecies had been reduced to such low numbers by alien mammals that they were ecologically extinct; that is, unlikely to perform a functional role in the island ecosystem (sensu Estes *et al.* 1989). We considered a taxon ecologically extinct if a competent researcher was unable to detect its presence after several visits; the majority of these extinctions can be attributed to alien mammals (Jehl and Everett 1985; Mellink 1992;

Smith *et al*. 1993; Howell and Webb 1995; Alvarez-Castaneda and Cortes-Calva 1996; Alvarez-Castaneda and Patton 1999; Collins 1999; Donlan *et al*. 2000; Junak and Philbrick 2000). Thus, as we use it here, ecological extinction is synonymous with possible global extinction. We chose ecological extinction since global extinction is often difficult to confirm on these remote islands and, in the short term, the ecosystem impacts are synonymous (Estes *et al.* 1989). Our assessment concentrated on vertebrates, since they are the most studied and well-known group in the region (Howell and Webb 1995; Alvarez-Castaneda and Patton 1999; Grismer 1999). We combined data for species and subspecies since both taxa are evolutionally significant (Ryder 1986; Rojas 1992), and the distinction between them is often dependent on how well and how recently the taxonomy for a given group has been revised. Some seabirds were driven to local extinction on one island, but populations persisted on other islands (McChesney and Tershy 1998); we recorded these local extinctions separately. All data on the distribution and impact of alien mammals were compiled in the aforementioned database for planning and research, which is accessible to the public for planning and research (Donlan *et al*. 2000).

Alien mammal eradications

Black rats (*Rattus rattus*) were eradicated from four islands (San Roque Island and the three San Jorge Islands), and Norway rats (*R. norvegicus*) and house mice (*Mus musculus*) from Rasa Island (Table 1). A bait station approach using rodenticide was employed in all cases (sensu Taylor and Thomas 1993). The bait stations were placed evenly across islands on a 25 x 25m grid. Extra stations were added along the shoreline where rat densities tended to be highest. On the San Jorge Islands, three rodenticides were used in the eradications. Brodifacoum (50 ppm, Final® Blox™ Bell Laboratories) was used on the main island, diphacinone (50 ppm, Ditrac® Blox™ Bell Laboratories) on the east islet, and cholecalciferol (750 ppm, Quintox® Bell Laboratories) on the west islet. Brodifacoum and diphacinone bait were in 20g extruded cereal wax blocks. Cholecalciferol bait was in cereal pellet form and dispensed in 10g packages. Bait stations remained active for one year; details of the San Jorge eradication are discussed elsewhere (Donlan *et al*. in press). On San Roque Island, brodifacoum wax blocks were used in combination with 100 ppm bromethalin in a gel bait; stations remained active for one year (Donlan *et al*. 2000). In 1994, Norway rats and house mice were eradicated from Rasa Island by Jesús Ramírez (deceased) of the Instituto de Ecologia, using bait stations on a 25 m grid containing 50 ppm brodifacoum wax blocks. None of the authors were directly involved in the Rasa Island project.

Successful eradication of introduced cats, rabbits, goats, and sheep was accomplished through a combination of environmental education and hunting and/or trapping. Hunters tended to work simultaneously on several islands at any one time, moving opportunistically between the islands depending on the number of alien mammals that appeared to remain on the island and logistic factors such as weather, transportation, and the availability of rifles and ammunition.

On Natividad Island, community education programmes resulted in live removal of sheep, goats, and dogs by island residents (Keitt 1998; Donlan and Keitt 1999). With active eradications, rabbits and cats were hunted both day and night (often with the aid of trained dogs), and trapped with Victor # 1½ padded leg-hold traps. Cat hunting and trapping techniques are described in detail by Wood *et al.* (2002). Rabbits were hunted during the day and night with 12 gauge shotguns and .22 calibre rifles. Dogs, Jack Russell Terriers, were used to hunt rabbits primarily during the day. Typically, a hunter would follow a single dog from a distance of up to 200m, often simply watching or listening to the dog from an elevated vantage point. Only when the dog's behaviour suggested it had located a rabbit, would the hunter investigate the area in detail. Jack Russell Terriers were often able to locate hidden rabbits and crawl into holes to reach them. However, hunters usually set traps outside occupied rabbit holes, so they could be more certain they had captured the occupant.

Goats were removed by hunting during the day with .22 and .222 calibre rifles. Dogs were not used for goat hunting. All hunting and trapping were done on foot, but small boats were sometimes used to move hunters/trappers to different parts of the islands.

After each island was thought to be free of the target species due to the absence of sign, at least two subsequent visits were made at three to eight month intervals to check for new sign. Only if no fresh sign was found on these subsequent visits, was an eradication considered to have been successful.

RESULTS

Distribution and impact of introduced mammals

In 1994 alien mammals occurred on at least 44 islands in north-west Mexico and are implicated in causing the ecological extinction of 22 endemic species and subspecies of vertebrates, as well as the local extinction of one or more seabird species from at least 10 islands (Fig. 1). Of the 22 ecological extinctions, there is substantial evidence that some have suffered global extinction (Mellink 1992; Smith *et al*. 1993; Howell and Webb 1995; Alvarez-Castaneda and Patton 1999).

Introduced mammal eradications

In collaboration with The Instituto de Ecologia at the Universidad Nacional Autonoma de Mexico, Centro de Investigaciones Biologicas del Noroeste, the national and regional offices of Areas Naturales Protegidas, and local people and community organisations, we have eradicated

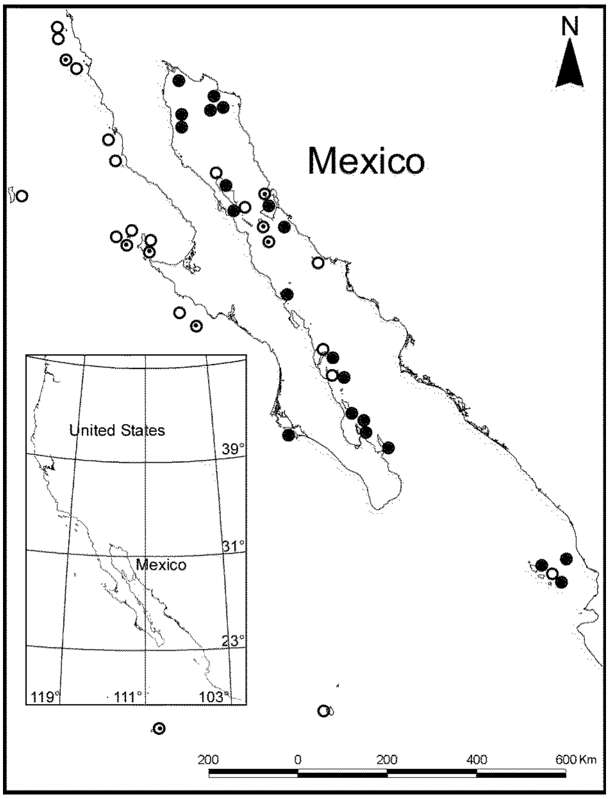

Fig. 1 North-west Mexico islands with alien invasive mammals in 1994 that did not suffer ecological extinctions (black circles), that suffered one or more ecological extinctions of endemic taxa (circles with black dot), or suffered the local extinction of one or more seabird species (white circles).

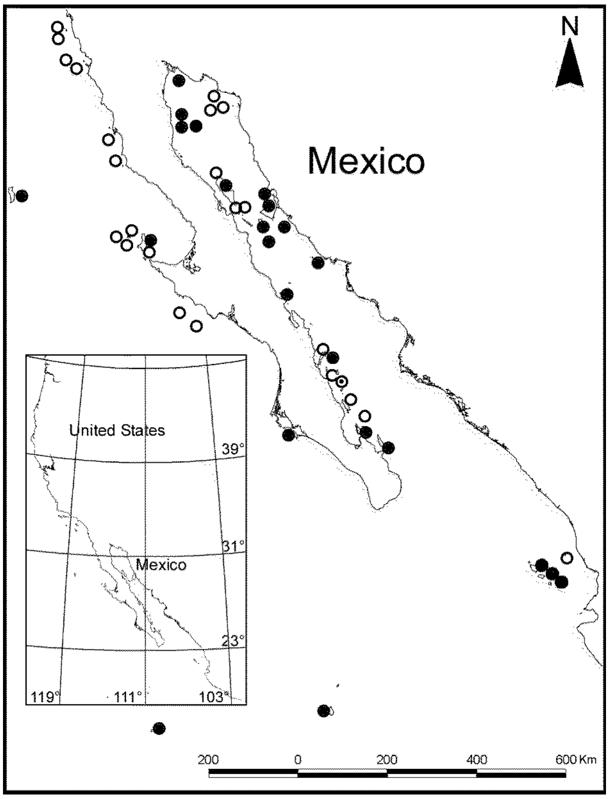

Fig. 2 The eradication of one or more invasive alien mammals from 23 islands in north-west Mexico. Islands from which one or more alien invasive mammals have been removed (white circles), where eradication is nearly complete (circle with black dot), and where they remain (black circles).

one or more introduced mammals from 23 islands (Fig. 2, Table 1). Eradication on one additional island is near completion (Fig. 2, Table 1). This work has protected habitat for 38 endemic taxa of terrestrial vertebrates, and 27 breeding seabird taxa, seven of which are endemic to north-western Mexico (Table 1). The estimated cost of these conservation actions, excluding the work on Rasa Island, was approximately USD750,000.

The first eradication was conducted in autumn 1994 and the most recent completed eradication was finished in winter 2000. From the start of hunting and trapping to when the last animal was captured lasted anywhere from 24 hours (14 cats on <1 km^2 San Geronimo Island) to over nine months on San Benito Oeste (3.5 km^2), where approximately 400 rabbits were removed.

DISCUSSION

Islands are critical for the conservation of global biodiversity, and the islands of north-west Mexico are no exception (Velarde and Anderson 1994; Ceballos *et al.* 1998; Donlan *et al.* 2000). As in other parts of the world, the main threats to these island ecosystems are the predation, competition, and habitat alteration caused by invasive alien species (Mellink 1992; Smith *et al.* 1993; Velarde and Anderson 1994; Alvarez-Castaneda and Cortes-Calva 1996; Keitt 1998; Donlan 2000; Donlan *et al.* 2000). Alien mammals appear to be responsible for more than 90% of the ecological extinctions of endemic vertebrates, and numerous local extinctions of seabirds (Donlan *et al.* unpub. data; McChesney and Tershy 1998). Fortunately, due to techniques developed by New Zealand conservation practitioners (Towns *et al.* 1990), alien mammals can be removed from islands in this region as evidenced by the successful projects described above.

These conservation successes were made possible by the model that we developed: an integrated bi-national team that coordinates and facilitates all aspects of island conservation (applied research, prioritisation, fundraising, public support through community education, alien mammal eradication, and protection against new introductions). Our research programme has provided evidence for population and ecosystem-level impacts of invasive alien species and insight on the recovery of systems after mammal eradication (Keitt 1998; Donlan 2000; Donlan *et al.* 2002; Roemer *et al.* 2002). Research at the regional level, particularly the development of the conservation database, has provided a biogeographical framework to prioritise our conservation efforts, as well as providing a conservation tool to Mexican government agencies (Carabias-Lillo *et al.* 2000; Donlan *et al.* 2000). A bi-national framework allows access to U.S funding opportunities, through our

Table 1 Islands from which alien invasive mammals have been removed and number of native taxa protected.

Islands (north to nouth)	Area (km^2)[1]	Aliens Removed	Breeding Seabirds[2]	Endemic species and subspecies[3]		
				Reptiles	Landbirds	Mammals
Pacific						
Coronado Norte[4]	< 1	Cats	11 (3[9])	2	2	1
Coronado Sur	1.8	Goats	7	4	2	1
Todos Santos Norte	< 1	Cats, Rabbits	5 (1[9])			2 (1[9])
Todos Santos Sur	1.0	Cats, Rabbits	6 (1[9])	2	1[9]	2 (1[9])
San Martin	3.2	Cats	6 (3[9])	3		2 (1[9])
San Geronimo	< 1	Cats	5			1
San Benito Oeste[5]	3.5	Rabbits, Goats	10	1	3 (1[9])	
San Benito Medio	< 1	Rabbits	10	1	2 (1[9])	
San Benito Este	1.1	Rabbits	12	1	3 (1[9])	
Natividad[6]	7.2	Cats, Goats, Sheep	6 (1[9])			1
San Roque	< 1	Cats, Black rats	6 (1[9])			1 (1[9])
Asuncion	< 1	Cats	7 (4[9])			
Gulf of California						
San Jorge East	< 1	Black rats	8 (2[9])			
San Jorge Middle	< 1	Black rats	8 (2[9])			
San Jorge West	< 1	Black rats	8 (2[9])			
Mejia	3.0	Cats	3	2		2 (2[9])
Estanque	< 1	Cats	1	1		
Rasa[7]	< 1	Norway rats, Mice	4			
Coronados	8.5	Cats	1	1		3 (2[9])
Monserrate	19.4	Cats	2	2		2 (2[9])
Catalina (incomplete)	43.1	Cats	2	8		1
San Francisco	2.6	Cats, Goats	1	2		2
Partida South	20.0	Cats	0	3		1
Isabela[8]	1.0	Cats	10			
TOTAL		32 removals	139 (27) 7[2]	33 (27)[3]	13 (6)[3]	22 (19)[3]

[1] Areas are estimates based on literature.
[2] 139 seabird populations (27 seabird species and subspecies), seven endemic to north-west Mexico.
[3] Number of endemic populations (number of endemic species and subspecies), some taxa occur on more than one island.
[4] Feral donkeys present.
[5] Donkeys are corralled and fed imported pelletised food.
[6] All feral dogs, and most pet dogs, have been removed, <10 pet dogs remain in the fishing village and residents have agreed to remove them by 2003; the ground squirrel *Ammospermophilus leucurus* was introduced from the adjacent mainland and is established.
[7] Project conducted by Jesús Ramírez (deceased) of the Instituto de Ecologia, Universidad Nacional Autónoma de México without participation of Island Conservation and Ecology Group.
[8] Island Conservation and Ecology Group assisted Cristina Rodríguez of Instituto de Ecologia, Universidad Nacional Autónoma de México; Norway rats still present.
[9] Possible extinctions (extirpations for seabirds) (e.g., 3 (2[9]) = three endemics, two of which may be extinct).

U.S office; these conservation dollars can be directed toward projects in Mexico where funding opportunities are less. Our Mexican branch facilitates efficient and successful interactions with Mexican government agencies, as well as develops local capacity. Local support of island users through community involvement and education in conjunction with eradications is critical, particularly with respect to the prevention of new introductions (Keitt 1998; Donlan and Keitt 1999).

We believe that this regional island conservation model is more effective than a series of single-island efforts for three reasons. First, the process of planning, obtaining public support, fundraising, and staff training does not have to be repeated for each island, and the knowledge accumulated during the course of each project is not lost when each island project is completed. Second, economies of scale enable an expert national or regional team to train and employ individuals with complimentary expertise who can, when appropriate, train or supervise a team of talented locals on each island or island group. Third, a regional perspective facilitates the selection of project islands based on an objective evaluation of team capacity, the available funding, the biological importance of the island, and the political and technical difficulties inherent in project.

Using this approach, in collaboration with our colleagues, we have removed alien mammals from most of the islands in north-west Mexico smaller than 40 km^2. With the experience and infrastructure developed on these islands, and the help of experts in New Zealand and other parts of the world, we hope to facilitate the removal of alien mammals from most of the remaining islands in north-west Mexico.

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A history of ground-based rodent eradication techniques developed in New Zealand, 1959–1993

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Abstract Eradicating rats from islands was for decades deemed highly desirable but considered practically impossible. This paper documents the development of ground-based rodent eradication techniques using bait stations in New Zealand up until 1993. The work culminated in a successful operation to eradicate Norway rats (*Rattus norvegicus*) from 3100 ha Langara Island in the Queen Charlotte Islands, Canada, in 1995.

Keywords Eradication; rat, *Rattus*; islands; bait station; rolling front.

INTRODUCTION

Introductions of rodents to new regions during centuries of exploration and colonisation around the globe are recognised worldwide as a major conservation problem (Atkinson 1985). New Zealand ecosystems developed without terrestrial mammalian predators, the only land mammals being native bats. The flora and fauna that evolved through long oceanic isolation during the Tertiary were vulnerable to the depredation of introduced mammals, and many of the extinctions that have occurred in this country can be attributed to the introduction of rodents (King 1984).

The first of the four rodent species introduced into New Zealand was the Pacific rat (*Rattus exulans*), which arrived with Maori, perhaps up to 2000 years ago (Holdaway 1996). This rat was transported widely around the Pacific by Polynesian peoples (Wodzicki and Taylor 1984) who utilised it as food. There were likely to have been intentional liberations that, along with natural dispersal, resulted in the species becoming widespread on the main New Zealand islands, as well as establishing on many offshore islands (Atkinson and Towns 2001).

One of the first documented records of Eurasian rodents in New Zealand is the account given by Anders Spaarman, a naturalist with Captain Cook, who described rats (probably Norway rats – *Rattus norvegicus*) coming ashore at Pickersgill Harbour when the *Resolution* was beached for careening in Dusky Sound, Fiordland, in 1773 (Rutter 1953). During the 19th century, the ship rat (*Rattus rattus*) and house mouse (*Mus musculus*) also became established in New Zealand (Atkinson 1973; Taylor 1975, 1978), probably accidentally as ship visits to the new colony of New Zealand increased in the mid-19th century.

The introduction of rodents has had a significant impact on native animal and plant species. What was accepted as unavoidable by colonisers of the day has been rued by naturalists, scientists, and conservationists ever since. In the 25 years since serious consideration was first given to the possibilities of rectifying this major ecological problem, eradication of rodents from islands has become an accepted conservation management tool – now used with much success in various parts of the world.

We describe here the history and development of ground-based, rodent eradication operations using bait stations in New Zealand, which led to the successful campaign to eliminate Norway rats from Langara Island in the Queen Charlotte Islands (Haida Gwaii) in Canada – at over 3100 hectares, the largest rat eradication achieved to date.

1959 TO 1976: TENTATIVE BEGINNINGS

With the establishment in New Zealand of various Government wildlife and science agencies in the mid-1900s, a better understanding of the distribution of rodents in New Zealand began to emerge (Wodzicki 1950; Watson 1956, 1961), together with a greater appreciation of the detrimental effect these predators were having in our ecosystems. A graphic example was the devastation and extinctions caused by ship rats on Big South Cape Island (Taukihepa), in the early 1960s where several locally endemic birds were extirpated (Bell 1978). Removal of rodents seemed insurmountable and understandably efforts were focussed on ongoing methods of control, rather than eradication.

For example, depredation of white-faced storm petrels (*Pelagodroma marina*) by Norway rats was noted in 1959 on Maria Island (1 ha) in the inner Hauraki Gulf (Fig 1). During the 1959 to 1961 breeding seasons, Mr A McDonald and other members of the Forest and Bird Protection Society, assisted in part by Don Merton and a £5.00 grant from Wildlife Service for poison, endeavoured to protect white-faced storm petrels on Maria Island and the adjacent David Rocks (less than 1 ha). This attempt used the warfarin-based rodenticide 'Rid-rat', which was distributed around petrel colonies (Merton 1961, 1962). From later visits in the mid-60s it appeared that rats had almost certainly been eradicated from each of these small islands (Moors 1985).

During the early 1970s, research on burrowing seabirds on Whale Island (Moutohora) in the Bay of Plenty included studies of the effects of Norway rats on grey-faced petrels

Pages 301-310 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

(*Pterodroma macroptera gouldi*) and sooty shearwaters (*Puffinus griseus*) (Imber 1978). A rat control programme was carried out over a small, 6.3 ha area of low-density seabird burrowing on the island, in which 800 (4 oz) packets of the warfarin-based poison "Prodide" were distributed, reducing rats to low numbers. However, re-invasion from outside the study area soon began to occur (Bettesworth 1972; Imber 1978).

Similarly, Norway rats were believed to be having a detrimental effect on breeding sooty shearwaters and flesh footed shearwaters (*Puffinus carneipes*) on Titi Island (32 ha) in the Marlborough Sounds. In the belief that total eradication was unachievable, Brian Bell (1969) suggested to the Lands and Survey Department that a control programme be put in place to reduce rat numbers during the chick fledging period. In December 1970, Dick Veitch of the Department of Internal Affairs, Wildlife Branch, laid 310 4oz packets of "Prodide" at about 15 m intervals along or near the main ridge of the island. Subsequent checks indicated that the rat population had been significantly reduced (Veitch 1970, 1971). Follow-up was intermittent and rats continued to be recorded. There was further poisoning around the colonies in December 1973, but checks by Lands and Survey personnel in March 1975 indicated rats were still present. Another application of poison was undertaken in May 1975, but there was no follow-up monitoring until we visited in 1981–1982. Our intention was to use Titi for an experimental eradication of Norway rats, but after continuous trapping over a six month period (approximately 9000 fenn trap-nights) we found rats were no longer present (Gaze 1983).

Despite widespread concern at the possibility of rats reaching rodent-free islands, it was not until November 1976 that a concerted effort was made to address the problem. On advice from Ian Atkinson, Rowley Taylor and Brian Bell, members of the Outlying Islands Committee, and at the urging of John Yaldwyn (National Museum of New Zealand), a symposium on the 'Ecology and Control of Rodents in New Zealand Nature Reserves' was organised by the Department of Lands and Survey (Coad 1978). This conference brought together over 50 of New Zealand's rodent researchers, wildlife practitioners and managers of island reserves – representing many government departments, research organisations, museums, and universities.

Presented papers and discussion ranged widely from the effects of rodents on ecosystems to the possibilities for control and eradication. It is indicative of general thinking of the day that despite a report of the apparent eradication of rats from Maria Island and the David Rocks (we did not know the outcome for Titi Island at this stage), in a final comment the Chairman, John Yaldwyn, concluded: "*We have control methods, and methods for reducing populations, but complete extermination on islands is remote or at least a very very difficult thing indeed.*"(Yaldwyn 1978). Nevertheless, the meeting overall provided the impetus for several individuals to pursue their ideas to develop methods for eradication of rodents.

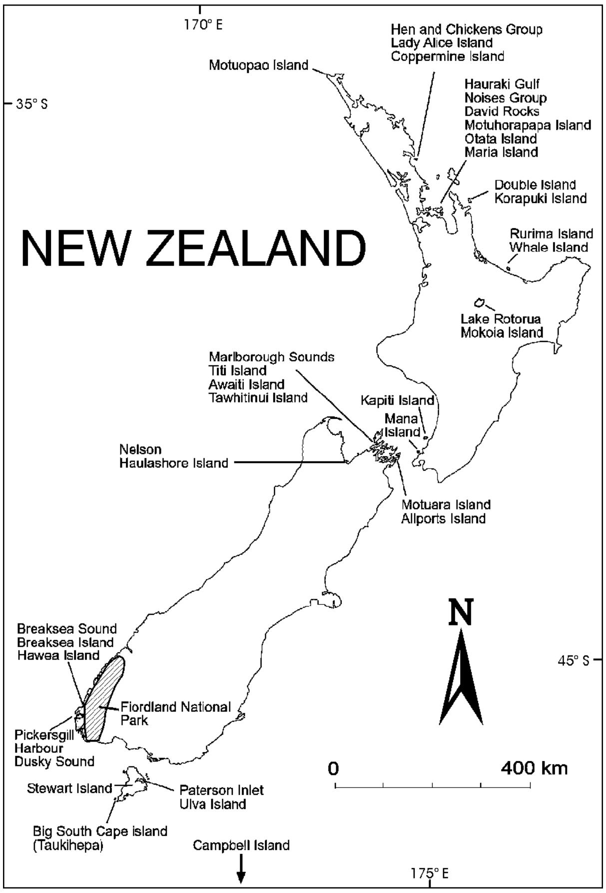

Fig. 1 Island localities mentioned in text.

1977 TO 1984: EARLY RESEARCH

Although most rodent research in New Zealand continued to be directed towards the distribution and ecology of rats and mice (Wildlife Research Liaison Group 1984), work now began on developing rat eradication techniques for islands. This was aided by the production of new and improved toxins in the form of second generation anticoagulants in proprietary rodenticides such as 'Talon' (brodifacoum) and 'Storm' (flocoumafen).

Recognising the difference in approach needed between eradication and control operations, Phil Moors of the New Zealand Wildlife Service began to test the feasibility of eradicating Norway rats from islands by undertaking differing poison trials on Motuhoropapa (9.5 ha) and Otata (21.8 ha) islands in the Noises Group (Moors 1978, 1979). It was thought eradication had been inadvertently achieved on Motuhoropapa in 1977–1978 as a result of his preliminary trapping study, and Moors postponed his planned poison programme to test this result. However, monitoring revealed that rats were still present in low numbers and the poison programme was reinstated in 1981. This involved a combination of compound 1080 (sodium monofluoroacetate)-impregnated grain, distributed in 75 plastic bait stations placed at 50 m intervals along existing tracks, plus 1080 paste spread in likely haunts around the coast and on the offshore stacks. A few months later, the 1080 baits were replaced with 0.005% brodifacoum 'Talon'

WB 50 waxed baits and 0.01% brodifacoum paste (Moors 1985). Kill traps (over 1400 trap-nights) were also set. No rats or sign were found after February 1983.

In 1979, bait stations were established on Otata Island (21 ha) on a grid at 40 m spacings. Poisoning was undertaken in two stages, the first using Compound 1080 in a mixture of rolled oats and fish-flavoured cat food, little of which was touched. Single ‘Talon’ WB 50 baits (about 7500) were then placed at 10 x 5 m spacings over the island, but a second island-wide poisoning campaign was deemed necessary in September 1980 after rodent droppings were found (Moors 1985). Eradication was confirmed on both these islands in 1987 (Veitch and Bell 1990). Although this work eventually met with success, it required considerable time and effort and led Moors (1985) to recommend: “*use as many methods of killing rats as you can, and never rely on one weapon alone*”.

Stemming from this work, Ian McFadden (1984) developed and tested bait stations, dispensing silos and various forms of baits and attractants on Pacific rats on Lady Alice Island. This technology was refined during eradication trials of Pacific rats in 1983–1984 on Rurima Island (6 ha) in the Bay of Plenty. After pre-feed trials, he used 1080-impregnated kibbled maize dispensed from 30 gravity-fed silos, but the rats did not take the bait (probably because of the taste of dyes used in manufacture). Subsequently, undyed kibbled maize impregnated with the anti-coagulant bromadiolone was used in an expanded array of silos (41 in total). Rodent feeding sign at the silos and monitoring using snap traps, indicated that eradication was probably achieved within three months of laying the poison (McFadden and Towns 1991).

This successful method was applied on Korapuki Island (18 ha), where McFadden improved his silo methodology and eradicated Pacific rats after just one application of bromadiolone-impregnated toxic kibbled maize (McFadden and Towns 1991). McFadden’s next experiment on Pacific rats, on Double Island in 1989, compared the cost and effectiveness of bromadiolone-poisoned grain in silo bait stations on one half of the island against hand broadcast distribution of commercially-available flocoumafen-based “Storm” rodent pellets on the other half (McFadden 1992). Both techniques achieved successful eradication, but the cost of broadcasting baits was marginally cheaper. Because of potential cost savings McFadden explored the development of aerial broadcast technology in which poison is distributed from spreader buckets slung under helicopters (McFadden and Green 1994). It is an eradication technique now widely used in New Zealand that has proved especially effective in difficult terrain and/or isolated situations, and with which success is being achieved on larger and larger islands (Cromarty *et al*. 2002).

While Wildlife Service was undertaking their early work on northern islands, the Department of Scientific and Industrial Research (DSIR) Ecology Division, was engaged in a series of rodent distribution surveys – involving Bruce Thomas (BWT) and others – in the Nelson/Marlborough region under the leadership of Rowley Taylor (RHT). Through this work (Taylor 1984) links were developed with the Department of Lands and Survey, who in 1980–1981 were considering rat eradication trials on Campbell Island. RHT recommended that, given current knowledge, a more appropriate plan would be for Ecology Division to cooperate in trials of rat eradications on smaller, readily-accessible islands in the Marlborough Sounds.

In planning the trials, we adopted a different approach from that recommended by Moors (1985). We tested a single-hit, single-poison methodology, taking account of the known behaviour of rats. Our aim was to develop a system of dispensing a proven rodenticide into the territory of every rat on an island, in a way that would minimise non-target poisoning, and monitor the effectiveness of the campaign as it progressed (Taylor and Thomas 1989).

In a joint DSIR Ecology Division/Marlborough Sounds Maritime Park programme, David Taylor and two other Lincoln College students, under Rangers Dave Maizey and Bob Ryan, carried out the fieldwork for the first two trials. The initial experiment was against ship rats on Awaiti Island (2 ha). Simple bait stations with a top-loading access slot and a clip-in cover were made from 65 mm diameter plastic drainage pipe (‘Novacoil’). About 120 bait stations were sited approximately 15–20 m apart over the island and a single 15 g “Talon WB 50” pellet containing 0.005% brodifacoum, a second generation anticoagulant rat poison, was placed in each tunnel. Pellets were replaced as necessary during weekly checks (i.e. up to five times between 10 March 1982 and 16 April 1982) – from which time no further baits were taken. Follow-up monitoring (kill traps and tracking tunnels) confirmed eradication had been achieved (D. Taylor 1983).

The second trial also targeted ship rats on the adjacent, forested, Tawhitinui Island (21 ha). A network of tracks was cut to give access to the coast at regular intervals from along the main ridges of the island. A total of 374 bait stations of 65 mm plastic drainage pipe were placed along these tracks, achieving a variable grid of 25–50 m. A single “Talon WB 50” bait was placed in each of the tunnels and checked and replenished weekly from 26 January 1983 until the poison-take stopped on 15 February 1983. Poison baits were left in place until February 1984 and the campaign was considered a success in August 1984 after follow-up monitoring (baited tracking tunnels and snap-trap lines) detected no further sign of rats (D. Taylor 1984).

About the same time, a review of Wildlife Service research priorities gave the highest priority rating for new predator projects to “*The development of eradication methods for use on small islands*” (Crawley 1983). We were already convinced of the potential to further develop bait station technology as a rat eradication technique for much larger islands, but a general scepticism of this methodology persisted amongst administrators, researchers and wildlife practitioners. For example, in a priority listing of 11 research topics on rodents, the Wildlife Research Liaison Group (1984) gave top priority to mapping rodent distribution. By comparison, 'methods of eradication or control' were placed low on the list at priority 9, with specific reference made only to possible biological methods. Despite our success in the Marlborough Sounds, the use of rodenticides in bait stations did not rate a mention. In 1983, influenced by the problems that Moors was encountering on the Noises Islands, Ian Atkinson voiced the then-commonly-held view that *"Once rats have established on an island, it is generally not feasible to remove them unless the island is very small."* (Atkinson 1986).

The Department of Lands and Survey was keen to continue its support for research aimed at rat eradication on Campbell Island, and in 1983 DSIR Ecology Division negotiated a research contract to further this objective. Graeme Taylor was employed to study Norway rats on the island to assist planning for an eventual eradication campaign. Key aspects of his research were rat distribution, density, breeding, food, habits, and home range size (Taylor 1986).

1985 TO 1988: THE BREAKSEA SOUND RAT ERADICATION CAMPAIGNS

Norway rats were confirmed present on Breaksea Island (170 ha) and the then unnamed Hawea Island (9 ha) during an ecological survey of islands in Breaksea Sound in 1974 (Thomas 1975). The possibility of eradicating Norway rats from Breaksea Island to create a predator-free environment in which to translocate the last few Fiordland kakapo (*Strigops habroptilus*) was discussed in depth by the team members during the survey. Enthused by the 'Rodents in Reserves' symposium, and encouraged by the success of the Marlborough Sounds trials, during subsequent work for the Fiordland National Park Board, BWT formulated ideas to undertake further development of bait station rodent eradication technology in Breaksea Sound (Taylor *et al.* 1986).

In April 1984 BWT, RHT and Fiordland National Park staff were joined by the director of Ecology Division on another ecological survey of islands in Breaksea and Dusky Sounds (Thomas and Taylor 1988). Our director was less than convinced with our proposal to eradicate rats from Hawea and Breaksea Islands, believing that eradication of rats from an island as large and rugged as Breaksea was not achievable – a sentiment echoed time and again from many quarters. Undaunted, and with the support of Department of Lands and Survey colleagues, we gained a small grant and an offer of logistical support from the Fiordland National Park Board. This enabled us to finalise plans to undertake an experimental eradication operation against Norway rats on Hawea Island, with the clear intention of expanding the programme to include Breaksea Island should we be successful.

We believed that the single best method available should be employed to achieve eradication in the shortest possible time. In undertaking the Hawea Island campaign we hoped to develop existing technology further to overcome problems such as bait station design, neophobia, bait avoidance and poison resistance, and monitoring success – all of which had compromised previous eradication operations to various degrees. With the help of Graeme Taylor and the voluntary assistance of several other people (Taylor and Thomas 1986), a track system and a preliminary programme to monitor ecological changes following rat eradication was completed on Hawea Island in 1986. Seventy-three 100 mm diameter plastic drainage pipe bait stations, each 400 mm long, were placed on a 40 m grid over the island three weeks before poisoning, to minimise neophobic avoidance by rats. From 11-22 April 1986, two "Talon WB 50" baits were placed in each tunnel and checked and replenished daily – a monitoring regime that enabled collection of precise data on rat activity. Eradication was accomplished in less than two weeks, and the system was self-monitoring and required no special effort to get the last rat (Taylor and Thomas 1989).

We could now recommend with confidence that a similar poison campaign be carried out on 170 ha Breaksea (Thomas and Taylor 1988). A project proposal submitted to DSIR gained research support for a Breaksea Island campaign for the period 1987–1989 (Ecology Division 1987). However, for the programme to go ahead it was essential that our draft work plan (Taylor and Thomas 1987) be accepted by the newly-formed Department of Conservation (DOC). The plan required an eight bunk hut and two bivvies to be built, hundreds of person-hours cutting tracks, the production of up to 1000 plastic drainage pipe bait stations, and 500 kg of "Talon 50 WB" rat poison – over NZ$50,000 for materials alone. The new managers we were dealing with were reluctant to commit resources, unconvinced that we could achieve eradication on such a large scale. By then, based on McFadden's work, conventional thinking was that *"rodent extermination on islands up to 40 or 50 ha might be possible."* (Towns 1988).

Two Te Anau DOC staff, Tom Paterson and Ron Peacock, shared our vision and were instrumental in the project being designated an official "Fiordland National Park Centennial Year Project". They secured some old Ministry of Works buildings, which provided the materials for the huts, successfully negotiated with ICI (Imperial Chemical Industries, now Zeneca) to donate the poison, and organised for the participation of "Operation Raleigh". A commitment by DOC was made to go ahead with the project, and Ian Thorne took responsibility for coordinating preparation of the island (Department of Conservation 1988). In 1987, several teams of young people from New Zealand and around the world, paying for an outdoor adventure experience with "Operation Raleigh", spent weeks under canvas in harsh conditions on Breaksea Island marking routes and cutting tracks, which were completed by DOC staff and voluntary helpers.

The Breaksea Island campaign was similar to the Hawea Island poison operation, but stations were more widely spaced (50 m apart) along contour tracks cut at 60 m vertical intervals from the coast to the summit. Thus, the surface distance between lines varied from about 30-100 m depending on the steepness of the terrain. Extra stations were installed at 25 m intervals along the main access ridges, with all 743 bait stations in place two months before poisoning. Six large weather-proof stations, each containing 50 "Talon WB 50" baits, were positioned by helicopter on inaccessible cliffs and offshore stacks. During the main poisoning operation (26 May to 16 June 1988), stations were loaded with two "Talon WB 50" baits and checked and replenished daily. Six poison operators, led by Ian Thorne, each had responsibility for a section of island. Bait-take was analysed daily to monitor the changing status of the rat population. As the operation progressed as predicted, even the sceptics in the team changed their views on our chance of success. On day 21, leaving the island loaded with four talon baits per station, we were confident that only two already-poisoned rats remained alive. Two years of post-poison monitoring confirmed our success (Taylor and Thomas 1993).

This created the largest predator-free island in Fiordland and advanced understanding of eradication technology. However, the most important outcome was improved confidence amongst administrators, conservation practitioners, politicians and the public alike, that eradication of rodents was achievable on a large scale – that money was not being squandered in attempting such operations. This was aided by raising awareness of the project through various media, the most important being the production of the Television New Zealand Wild South documentary 'Battle for Breaksea Island' (Natural History New Zealand Ltd 1990). This 26 minute television documentary, shown in New Zealand and overseas, has had a tremendous impact on predator eradication efforts.

1989 TO 1993: THE "ROLLING FRONT" AND OTHER CAMPAIGNS

The Breaksea Sound work was to us a preliminary step towards eradicating rats from Kapiti Island (Thomas and Taylor 1988), a 1970 ha island sanctuary of world renown and of particular importance to the conservation of several critically endangered bird species (Maclean 1999). Removal of feral stock had been achieved, and possums eradicated in 1986 (Cowan 1992; Maclean 1999). Pacific and Norway rats were the only introduced mammalian predators that remained and we felt the time was right to give consideration to their eradication. We prepared a discussion document proposing that because of the island's size a ground-based poison campaign be undertaken sequentially in three stages in what we termed a "rolling front" regime, and recommended that it first be tested elsewhere (Thomas and Taylor 1991). For reasons of size, shape, ease of access and the fact that it had Norway rats, we suggested that Ulva Island (270 ha) in Paterson Inlet, Stewart Island, would be the best place to undertake such trials. Southland Conservancy of DOC agreed that their existing programme for Ulva, based on methods used on Breaksea Island, should be modified and the island used to trial the rolling front on a "research by management" basis.

Responsibility for this project had been given to DOC officer Lindsay Chatterton who implemented the changes necessary to undertake the more complex "rolling front" programme. The island was divided into three blocks of 70–100 ha, which were to be poisoned sequentially. A total of 282 bait stations were placed along 47 lines to achieve a 100x100 m grid over the whole island. There were concerns about the possibility of unnecessary amounts of poison entering the food chain, and the effects of this on non-target species and the environment. To study optimum baiting levels, each block received a different loading of "Talon WB 50" poison, from an extremely low dosage in Block 1 to a dose somewhat less than we used on Breaksea Island in Block 3 (Fig. 2); and the check-replenishment regime was pulsed according to the particular stage of the campaign. Just before the poisoning began we learned that "Talon WB 50" baits had been in regular and widespread use on Ulva for rat control for over 10 years. This presented a possibility that the programme could be compromised by bait avoidance or poison resistance in the rat population (Taylor and Thomas 1989).

Poisoning began in Block 1 on 6 July 1992, with two Talon baits per station checked and replenished by two operators every two days. Three weeks later, poisoning started in Block 2 with four baits per station. A further four weeks later, in late August, eight baits per station were applied in Block 3. As the "rolling front" moved ahead into the next block, bait loadings were doubled and the frequency of checks reduced. On 24 October, all stations on Ulva Island were loaded with 10 baits and less frequent, but regular, checks continued until April 1993. Non-toxic indicator baits were also distributed over the island at this time and fenn traps set to monitor success and catch surviving rats. There were marked differences in results between blocks. Maximum daily bait-take peaked much earlier and declined more rapidly in high-dosage Block 3 compared with low-dosage Block 1, where the peak was delayed, the high bait-take period was protracted and the overall period of bait-take continued for longest.

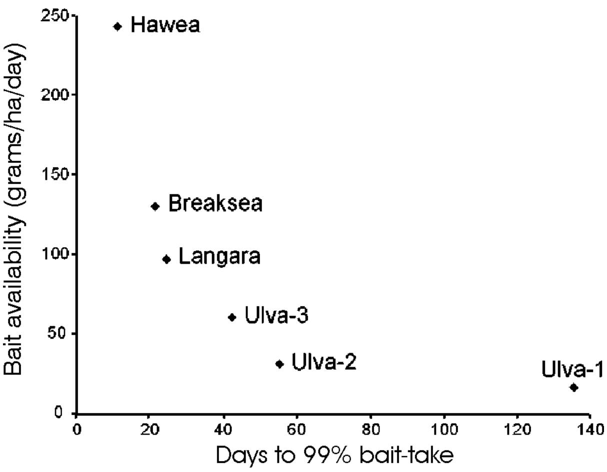

Fig. 2 Relationship between bait availability and duration of rat-eradication campaigns.

Our concerns at the possibilities of bait avoidance were also realised. A number of poison-resistant or bait-shy individuals survived on Ulva, after the rest of the rats had been poisoned. At least two females became pregnant several months after Talon bait was laid in their block, and one gave birth to at least three young before being finally trapped. Other trapped rats were found to have brodifacoum concentrations in their livers well in excess of that usually found in rats poisoned by Talon. The brodifacoum-resistant rats were detected through the monitoring regime, which included the use of non-toxic indicator baits as well as the self-monitoring nature of the bait-station method. Hotspots were eventually cleared by trapping (11 rats caught) and/or bromadiolone-poisoned crushed maize.

In response to concerns by some DOC staff that 100 m spacings of bait stations would be too wide, and to record responses of rats living adjacent to the poison fronts, 18 Norway rats were live-trapped in the vicinity of the poison fronts shortly before or during poisoning. Each rat was fitted with a radio transmitter, released and tracked. Of 11 transmitters used in the study, two failed after six and 18 days respectively and were never recovered, a male rat lost its transmitter after two days (probably through poor fitting), a female rat in poor condition was located dead the day after she was caught and, where appropriate, some transmitters recovered from poisoned rats were reused. Six of the transmitters had their aerials chewed off at the base but emitted strong enough signals for tracking to continue.

Rats were tracked for periods ranging from two to 69 days. Dens of marked rats were located daily and as many locality fixes as possible were obtained for each rat during their period of night-time activity. Single, linear movements of over 600 m in a night were recorded and all telemetered rats were recorded in the vicinity of one or more bait stations. As poisoning progressed and population pressure reduced, some rats moved up to 400 m into poison-activated areas from adjacent non-poison blocks. Most of the telemetered individuals died less than two weeks after Talon had been laid in their block, however, three of the study animals persisted for more than a month after baiting commenced. This pattern was quite different to what we had experienced in the Hawea and Breaksea campaigns, and a further indication that a percentage of the rat population on Ulva Island was either bait shy or poison resistant.

Concurrent with eradication operations on Hawea, Breaksea and Ulva Islands, important developmental work was also being undertaken in other areas by DOC workers. In 1989, Paul Jansen adapted the Breaksea Island work plan to eradicate Norway rats from Mokoia Island (135 ha) in Lake Rotorua (Veitch and Bell 1990). By 1990, bait station technology had been used to eradicate rodents from about 18 New Zealand islands, and a similar number had been cleared by hand broadcasting baits (Veitch 1994). Pacific rats were eradicated from Motuopao Island (30 ha), Northland, in 1990, and Motuara Island (59 ha), Marlborough Sounds, in 1991, with bait stations placed on a 50 m grid (McKenzie 1993; Cash and Gaze 2000). To address further the spatial requirement of bait stations for Pacific rats, an attempt was made to eradicate Pacific rats from Coppermine Island (80 ha) in the Hen and Chickens group using 100 m spacings. The operation failed but it was difficult to determine if this was because of the wider bait station spacing or the rugged topography of the island, which was later cleared of rats using aerial broadcast (McFadden 1997).

Spacing of bait stations was also a consideration on Allports Island (16 ha) in Queen Charlotte Sound in 1989, when Derek Brown (1993) successfully employed the Breaksea bait station system to eradicate mice using a 50 m grid. In exterminating mice from 217 ha Mana Island in 1989 a 25 m spacing of bait stations was used. However, an aerial application was also incorporated into this campaign and so a combination of factors brought about success (Hutton 1990; Hook and Todd 1992).

We felt throughout our trials that McFadden's concept of silo bait-dispensing technology was an alternative or adjunct to ground-based eradications (McFadden and Towns 1991) and we used "bulk bait stations" in the difficult areas at Breaksea. In 1991, the Nelson City Council approached us for advice on rats on Haulashore Island (6 ha), in Nelson Harbour. Index trapping indicated that this island had a population of approximately 300 ship rats, which we undertook to eradicate. This was a trial of low station density, with a weekly checking regime. Two steel 12 gallon drums placed on their sides were converted into simple poison stations by installing a hinged, lockable, access door on top, a 100 mm entry hole for rats 25 mm above ground level at either end, and a shallow wooden tray for holding baits. The drums were positioned about 250 m apart, each loaded with 100 "Talon WB 50" baits and checked and replenished weekly. In total, 755 baits were taken, about 2.5 baits per rat, and eradication was achieved over a period of 90 days with minimal operator input.

This approach worked on a temperate island with a simple habitat structure, where rats were scavenging for seasonal foods and beach flotsam. However, it is unlikely to be successful in complex habitats (i.e. tropical forests), with an abundance of year-round food. The results on Haulashore Island strongly suggested that ship rats, like Norway rats, were also susceptible to "peer pressure" in following each other to food sources, with the last rats extending their home ranges to encounter one or other of the two poison stations.

As a result of several years of trials and investigations into how best to deal with the presence of two rat species in a single eradication campaign, the Department of Conservation successfully eradicated Norway and Pacific rats from Kapiti Island in 1996 using helicopter broadcast of Talon 7-20 pollard baits. Offshore stacks were treated by either aerial or hand broadcast of baits and bait stations were used on three small adjacent Islands (Empson and Miskelly 1999).

1994 TO 2000: LANGARA ISLAND

With the more economical and simpler aerial poisoning operations gaining wider acceptance in New Zealand, opportunities for further testing the bait station approach were diminishing. However, ground-based rat eradication technology provided a viable alternative in situations where aerial broadcast was not feasible or actually prohibited by law. This assertion was tested in 1995 when the ground-based techniques developed in the Breaksea and Ulva Island campaigns were extrapolated to much larger Langara Island (3100 ha) in the Queen Charlotte Islands, British Columbia, Canada. The island had once been one of British Columbia's largest seabird colonies, with six species of burrow-nesting seabirds. Over a period of 30-40 years, Norway rats had exterminated five of these as breeding species and reduced the others from 200,000 to 14,600 breeding pairs (Harfenist 1994). Using funds from the litigation settlement following an oil spill from the tanker *Nestucca*, this project was managed by Environment Canada (Kaiser *et al*. 1997).

The Langara operation was huge, and new considerations such as different habitat type, vulnerable non-target species and the presence of permanent settlements on the island needed to be catered for (Taylor 1993). The island was divided into five working units, each with a camp, a supervisor, a cook, and a team of six bait station operators. With 100 x 100 m spacings, it required close on 4000 Breaksea-type bait stations to attain full coverage of the island. The whole island was poisoned simultaneously using a baiting protocol in which checks and replenishment of baits were undertaken every two days. Within 300 m of the shoreline, where concentrations of rats were greatest, tunnels were loaded with 12 baits per station. Stations over the rest of the island received six baits. Apart from a few "stragglers" in two areas of human habitation (one was trapped and the others quickly dealt with by providing extra baits), eradication of rats from Langara was essentially achieved in less than four weeks from the time the first bait was laid (Kaiser *et al.* 1997; Taylor *et al*. 2000).

DISCUSSION

The bait station rodent eradication technique is based on a strategy that takes into account the characteristics of second-generation anticoagulant poisons, the behaviour of the target rodents, and the island environment. It is designed to monitor its own progress, kill every rat or mouse within a selected timeframe, continually detect the presence of surviving rodents, limit the risk to non-target species, and overcome many of the problems often associated with "getting the last rat". This technique has led to many successes in rodent eradication, dealing with Norway, ship, and Pacific rats and house mice on scores of islands, up to 3100 ha worldwide. Experience shows that 100 m spacing between bait stations is adequate for Norway rats in temperate regions. In New Zealand, ship rats, Pacific rats, and mice have all been eradicated successfully using bait stations spaced 50 m apart. Not all operations have been straightforward, but most problems have been associated with human populations, non-target species gaining access to inadequately designed bait stations, and poison resistance. In a few populations where these poisons have been used for "controlling" numbers of rats and mice over a long period, poison resistance is continuing to be a problem – even with second generation anticoagulants - (Quy *et al*. 1995).

Considering all the campaigns in which we have been involved, from Awaiti Island (2 ha) in the Marlborough Sounds, New Zealand, to Langara Island (3100 ha) in Canada, it is evident that the under-pinning factor and key to success was the simplicity and self-monitoring nature of this ground-based technique. The duration of such eradications is related to bait availability in an exponential manner – the more bait available, the faster the rats are killed (Fig. 1). The scaling-up of the technique from Breaksea to Langara was straightforward and did not require any significant changes to the methodology. With such a large-scale operation, involving a great number of people, many inexperienced, there was plenty of potential for problems to arise on Langara. However, the success of the campaign within its predicted timeframe demonstrates the reliability and inherent robustness of the method.

Ground-based eradication techniques have an important part to play in many parts of the world, despite the present emphasis in New Zealand on aerial operations (Cromarty *et al.* 2002). In the United States, Canada, and some European nations, there is legislation regarding the use of rodenticides out-of-doors, and the broadcasting of vertebrate poisons from the air is prohibited. Already, there are limitations on the aerial sowing of anticoagulant rodenticides on the New Zealand mainland and on stocked islands. Such restrictions are likely to become more widespread in future. Most of the world's biodiversity occurs in tropical regions (Africa, Madagascar, Indo-Pacific, South America, etc.) where the indigenous inhabitants lack financial resources to fund aerial operations, whereas labour is less of a problem. Bait station techniques are also the only option on densely-populated islands. On sparsely-populated islands, or where very vulnerable non-target species have a proscribed distribution, a mix of both methods is often appropriate. There is a need for continued development of bait station design – in particular, improvements aimed at excluding non-target species and limiting the entry of toxins into the environment.

Island rodent extermination campaigns that 20 years ago were thought impossible are now being tackled with confidence – by both ground-based and aerial operations. Two important developments made this possible. First, the advent of new, potent and highly-palatable "second-generation" anticoagulant poisons, and second, the design of rodent eradication strategies to take advantage of these new poisons and our increasing knowledge of rat behaviour. However, one of the most important breakthroughs allowing eradication to progress has been psychological – the acceptance that the job can actually be done (Thomas and

Taylor 1988). It is perhaps ironic that eradication of Norway rats had been achieved unwittingly from 32 ha Titi Island during “rat control” operations that ceased in 1975, but through lack of monitoring this remained unknown for many years. If this success had been revealed earlier, especially at the time of the 1976 symposium, progress in the development of rat eradication techniques may have been accelerated. This highlights the need for adequate monitoring of control and eradication programmes.

Today, hundreds of rodent eradication campaigns have been carried out around the world, and others are in progress. Not all are successful. The main reasons for failure are that the best poisons are not always used, there is a super abundance of alternative foods, there are complications with non-target species, the effort is not sufficiently organised and sustained, or rodents are able to re-infest either from boats or by swimming. Before any campaign begins, the chances of re-infestation must be thoroughly assessed, and plans formulated and actioned to detect and counter future invasions.

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Early detection of invasive weeds on islands

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Abstract Early detection of new invasive alien plant species (weeds) allows for early control. This improves the chances of successful eradication and minimises the impact of such weeds on biodiversity. Both are imperative on islands with high conservation values. Searching for new weeds is particularly important where there are roosts of seed-dispersing birds, a history of garden cultivation, or where neighbouring islands are weedy. Success with detection and control of new weeds on New Zealand islands has been variable. Sometimes infestations have been found when there were less than 10 individuals allowing immediate eradication. More commonly, early sightings have not been followed by prompt action, and infestations have become more expensive to control. Where detection has been too late to eradicate the weed, conservation values have been compromised. This paper gives examples of detection and control of weeds on New Zealand islands and anticipates an improvement in both with the development of a systematic approach to weed surveillance.

Keywords Invasive alien plants; early detection; weed surveillance; weed control; weeds; islands.

INTRODUCTION

New Zealand is a weedy place – both on the mainland and on its multitude of islands (Buddenhagen *et al*. 1998). In a study of the invasive alien plant species (weeds) on over 200 of New Zealand's islands (176 offshore and 36 outlying islands 5 ha or larger), 63% of the offshore islands and 19% of the more remote "outlying" islands had one or more invasive weed species (Atkinson 1997). The incidence was highest for islands in the northern part of New Zealand (4.3 weeds per island) and lowest for outlying islands (0.9 weeds per island). Some weeds have only a transitory effect but many can modify the structure of natural communities on islands and disrupt succession (Towns *et al*. 1997). In all, Atkinson (1997) identified 94 species requiring some level of control on the islands in the study and recommended that the most cost-effective way to manage weeds on islands is to detect weeds early and eliminate them before they establish properly. A system to achieve this, both on islands and the mainland, is described in the Weed Surveillance Plan for the New Zealand Department of Conservation (DOC; Braithwaite 2000). The system prompts timely and accurate identification of new populations of invasive weeds (i.e. those that are newly naturalised or established in an area). This weed surveillance system emphasises the importance of finding new weed infestations early, when effective action is still possible and before the cost of control escalates and the weed infestation has compromised natural values (Fig. 1). The system captures weed sightings from deliberate searching *and* casual observations of both members of the public and reserve managers (Braithwaite and Timmins 1999, 2000). The Surveillance Plan is consistent with the IUCN guidelines for eradication and control of invasive species, viz. early detection and rapid action are the keys to successful, cost-effective eradication of new invasives (IUCN 2000).

regular surveillance	⋗	early detection	⋗	prompt action	⋗	successful eradication

Fig. 1 Weed surveillance: the ideal pathway to minimise costs and maximise biodiversity.

Globally, there has been insufficient ongoing monitoring in natural areas to detect infestations early (Mack *et al.* 2000). This has compromised our ability to eradicate invasive species (Myers *et al.* 2000). All cases of successful control of invasive plant species were initiated during the early stages of invasion (Macdonald *et al.* 1989). It seems that usually there is no recall once a species becomes established and begins to spread (Mooney and Drake 1989). The Weed Surveillance Plan aims explicitly to increase watching for weeds in New Zealand. The Plan recognises the need for different sorts of surveillance for different circumstances. Surveillance may focus on particular species or particular places. Those places may be vulnerable to invasion by weeds or they warrant searching because they are of high conservation value. The surveillance effort may be an active search of a site, but it is also possible to follow-up fortuitous sightings or use existing information (Braithwaite 2000).

Islands deserve weed surveillance attention because of their conservation value and their vulnerability to weeds. Some islands (e.g. the Poor Knights, Three Kings and the Kermadecs (Fig. 2)), support endemic plant species. Others have healthier populations of native species than are found on the mainland (e.g. milk tree (*Streblus heterophylla*) on Mana, Maud or Stephens Islands). Other islands are the breeding places for threatened fauna. Weeds can directly threaten these valuable assets.

Internationally, islands tend to have more invasives than mainland sites of similar area (Lonsdale 1999). Atkinson (1997) identified five major factors influencing the spread of weeds to New Zealand islands: (1) weed infestations on adjacent mainland or neighbouring islands, (2) source of weed propagules close-by, (3) location in the path of pre-

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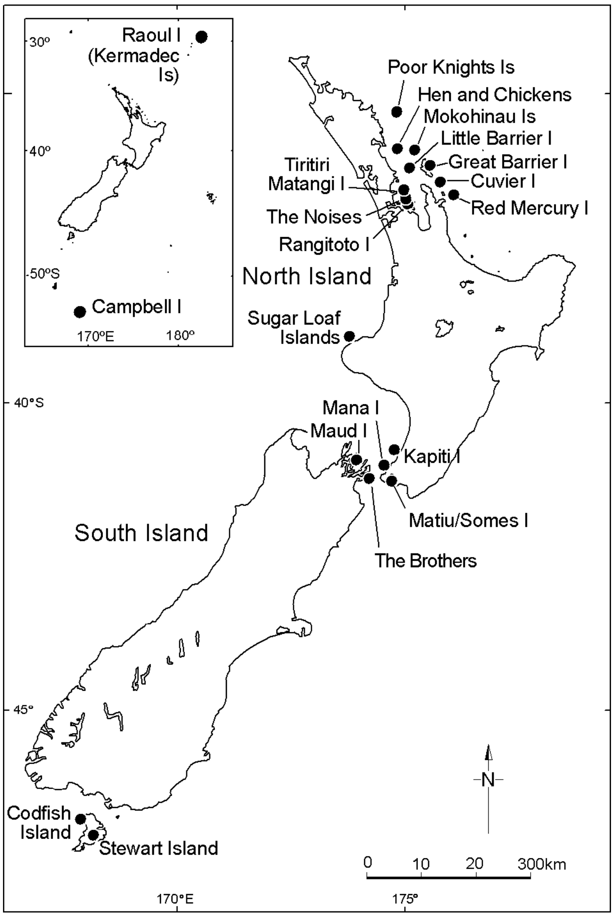

Fig. 2 Location of New Zealand islands mentioned in this paper.

vailing winds and potential weed sources, (4) roosts of the introduced starling (*Sturnus vulgaris*), and (5) history of garden cultivation.

Islands with these features are likely to be vulnerable to weed invasion and thus benefit most from regular weed surveillance. In a general sense, Rejmanek (2000) suggested that even a moderate increase in resources for early detection and eradication of invasive weeds would be a profitable investment. This has been confirmed for New Zealand. The appropriate frequency of such surveillance searches is a function of the biodiversity value of the island and its risk of weed invasion. A model for establishing surveillance frequency has been developed for mainland reserves (Harris *et al.* 2001).

Challenges

Finding a weed early is difficult enough on the mainland: islands pose extra challenges. Foremost among these is that islands are often difficult or expensive to get to. As a result, they may be visited infrequently, so that casual observations of new weeds are few. Also, island expeditions are invariably multipurpose and may be mounted in the wrong season for weed spotting. The dissected terrain of some islands also makes spotting and control of weeds difficult. Raoul Island, in the Kermadec group, provides a good example. A concerted effort has been made to control Mysore thorn (*Caesalpinia decapetala*), which is now uncommon there. However, the disturbed cliff sides cannot be safely inspected because they are too steep and dissected (West 2002).

As if the islands themselves don't provide enough challenges, weed surveillance of islands must also include checking and controlling weeds on adjacent mainland or island sites, particularly at take-off sites which are in line with the prevailing wind or the flight paths of bird-dispersers. For example, weed surveillance on Rangitoto Island (Fig. 2), near Auckland City, is complemented by control of weeds on nearby Motutapu Island, Browns Island, and North Head on the mainland. On Motutapu the species controlled include: barberry (*Berberis glaucocarpa*), boxthorn (*Lycium ferocissimum*), Chinese privet (*Ligustrum sinense*), hawthorn (*Crataegus monogyna*), monkey apple (*Acmena smithii*), Moreton Bay fig (*Ficus macrophylla*) and Port Jackson fig (*F. rubiginosa*), and on Browns Island Japanese spindle tree (*Euonymus europaeus*) is controlled to protect Rangitoto. None of these species are known to be present on Rangitoto (J. Wotherspoon pers. comm.). Another type of preventative action is checking for weeds in the plants brought to islands for restoration work. Plants brought to Mana Island and Matiu Island, near Wellington, are always checked for stowaway weed seeds or plants (Miskelly 1998).

The risk of weed invasions is further compounded by starling roosts or a long history of garden cultivation; both of which are common features of islands. Lighthouse islands have more naturalised plants than their unlit (no lighthouse) island neighbours because lighthouse keepers often established a garden and some of those garden plants escaped and spread. For example, Cuvier Island off Coromandel Peninsula (Fig. 2) has a lighthouse that was continuously staffed for 93 years (1889 to 1982). Today the flora of Cuvier Island is 35% adventive. This compares with Red Mercury Island, its unlit neighbour of similar size, whose flora is only about 12% adventive (J. Roxburgh pers. comm.).

The dilemma when finding an unknown plant species is exacerbated on an island. Is the unknown plant a weed or a threatened species? On an island the latter could well be true; if that proves to be the case, it would be distressing to have pulled it out. But if the plant proves to be a weed, how quickly can you get back to control it? This confusion occurred when white bryony (*Bryonia cretica* subsp. *dioica*) was 'discovered' in Makino Reserve near Wanganui (North Island of New Zealand). At first it was thought to be the threatened curcubit *Sicyos australis*. When it flowered and fruited the truth was revealed: it was white bryony, a bird-dispersed weed with massive tubers (C. Ogle pers. comm.). Although it had not been recorded in New Zealand before, the species had probably been in the area, at several locations but unrecognised, for a very long time.

Prior to the publication of the Weed Surveillance Plan, visitors to islands occasionally detected and reported new weed incursions. Sometimes the sightings were made by reserve managers doing weed work; sometimes by other staff such as those eradicating animal pests or translocating a threatened species. Some of these sightings were promptly acted upon; others were not. This paper reports examples of the mixed history of surveillance and action on islands in recent years, including both success stories as well as failures. For the most part we have collected our examples by talking with reserve managers working on islands.

SUCCESS STORIES: EARLY DETECTION AND PROMPT ACTION

Pampas grass (*Cortaderia selloana*) seed was introduced to Raoul Island (Fig. 2) when a retaining wall was built there. When pampas plants were spotted on the wall, they were promptly pulled out, as were a handful of adults in successive years (West 1996). The aim is to keep pampas grass off Raoul because it would readily colonise the open coastal faces and displace native plants.

Ragwort (*Senecio jacobaea*) on Raoul Island has a similar story. In 1980 a single plant was found near Mahoe Hut; perhaps seed came in with building materials for the hut. It too was pulled out at once and although the site has been checked regularly since, no further ragwort plants have been seen. On Raoul, ragwort could flower year round and thus it could have readily colonised the coastal slopes (West 1996). In both instances prompt action averted these exotic colonists establishing on Raoul's coastal cliffs (Fig. 3). In 1998, a sharp-eyed weeder on Raoul Island found selaginella (*Selaginella kraussiana*) (West 2002). It was sprayed soon after it was positively identified, but it keeps emerging at the same site. The site now has a sign alerting the presence of selaginella and any visitors are required to clean their boots in hot soapy water after visiting the site to prevent further spread of selaginella on Raoul (A. Warren pers. comm.).

Fig. 3 Raoul Island – regular weed surveillance is essential to keep the coastal cliffs free of new weed colonists (Photo: Rachael Barker).

On the subantarctic Campbell Island (Fig. 2), Colin Meurk spotted lotus (*Lotus pedunculatus*) around the old meteorological station in Tucker Valley. It was sprayed, and regular checks (1976–1996) have not detected it since (e.g. Meurk 1989). Another success story comes from the Hen and Chickens Islands south of Whangarei (Fig. 2). A small infestation of needlebush (*Hakea sericea*) was found in 1996 and the plants were pulled out immediately. They had already set seed, so it took a further two seasons of pulling out seedlings to eradicate it (G. Coulston pers. comm.). Banana passionfruit (*Passiflora mollissima*) on Kapiti Island, near Wellington is a similar story. When found seven years ago it was controlled promptly, so now there are no adult plants of this weed left on the island. A few juveniles grow each year from a seedbank that is likely to exist for some time but they too are promptly pulled (Russell *et al*. 2001). Mist flower (*Ageratina riparia*) was first recorded on the Poor Knights in 1991 and control started in 1994. Prompt action meant that this species was eradicated (G. Coulston pers. comm.).

In 1999, climbing dock (*Rumex sagittatus*) was found at one bay on Maud Island in the Marlborough Sounds (Fig. 2). The spotter was a member of the team managing the endangered kakapo (*Strigops habroptilus*). The infestation is now under control with a good chance of containment – a situation made possible by reporting a casual observation of a plant that looked a bit different (M. Newfield pers. comm.).

Two stories from Stewart Island (Fig. 2) are not such clear-cut successes. German ivy (*Senecio mikanioides*) was found in September of one year and treated the following summer – now all that is required is yearly checking and control of any regrowth at that site, but a further site has been found. When a reserve manager found selaginella, not previously known on Stewart Island, the first step was an advertising campaign, including talking to the next local Garden Group meeting. This identified several more infestations, also all on private property. One site was treated the following summer but more control, and a full survey, is needed to meet the aim of eradicating selaginella from Stewart Island (C. Wickes pers. comm.).

FAILURES: TARDY DETECTION AND INEFFECTIVE CONTROL

Prior to the release of the Weed Surveillance Plan, detection of weeds was more *ad hoc* and prompt control did not always follow detection. There were, and are, several stumbling blocks to effective surveillance. The most fundamental is not detecting the new weed incursion – because searching is too infrequent, in the wrong season to spot the weed, or the terrain obscures the searcher's vision. Even when a new incursion is spotted, sometimes the species is not recognised as a weed and thus not reported; or the incursion is reported but there are insufficient resources to attack it immediately; or the incursion is larger than first thought. Perhaps most frustrating is when a new incursion is spotted and eliminated, but the species continues to re-invade. The following examples illustrate some of the costs of delayed detection and/or control of invasive weeds.

In 1998 a large infestation of moth plant (*Araujia sericifera*) was found on Cuvier Island. It was removed, but the next year an extraordinarily thick carpet of seedlings appeared. Despite five re-treatments of the dense seedling mat, seeds still continue to germinate. It will be hard to eradicate moth plant from the island because it was already well established when it was found (J. Roxburgh pers. comm.). Similarly, moth plant was not recorded on Hen and Chickens Islands until 1996 but by then the main infestation was already 0.2 ha (G. Coulston pers. comm.). Delayed detection, plus continual re-invasion from the mainland, means eradication is unlikely.

By contrast, on Lady Alice Island delays in taking management action against Mexican devil (*Ageratina adenophora*), mist flower and pampas grass make it unlikely that these can now be eradicated (G. Coulston pers. comm.). Mexican devil at least has been known on this island for some years (Cameron 1984). For holly-leaved senecio (*Senecio glastifolius*) on Mana Island near Wellington, the stumbling block was frequency of surveillance. Although the weed was found and pulled out during a botanical survey of Mana Island in the 1980s, subsequent searching for this species on the island has been too infrequent so now the weed is quite widespread (Sawyer pers. comm.; Timmins *et al*. 1987). Conducting surveillance often enough is even more difficult for remote islands.

Sometimes new weed incursions have been found in good time but control has been tardy or sporadic. Evergreen buckthorn (*Rhamnus alaternus*) was first recorded on Rangitoto Island, on the summit, in the 1920s (J. Wotherspoon pers. comm.). It would have been much easier to control that single infestation than the now-dense coastal fringe of evergreen buckthorn that displaces native coastal shrubs and herbs. Similarly, mile-a-minute (*Dipogon lignosus*) was found on Rangitoto Island in 1990, but not controlled, so now there are several persistent patches of this weed on Rangitoto. A similar tale can be told for two grasses on other islands. Veld grass (*Ehrharta erecta*) was reported on Kapiti Island in 1982 (Ogle 1988). No action was taken so now this grass, which is shade tolerant, is widespread along Kapiti's coast and also spreading into the forest (Colbourne pers. comm.). It forms mats that overwhelm low-growing native plants and outcompete tree seedlings (Ogle 1988). It would be difficult to eradicate as it produces abundant seed for much of the year. Pampas grass was found on Cuvier Island in 1993 but not until 1998 were there resources to start control. By then it was firmly established and the chance to keep the cliffs free of pampas grass had probably been lost (J. Roxburgh pers. comm.).

Mouse-ear hawkweed (*Hieracium pilosella*) has been managed sporadically on Codfish Island off Stewart Island for many years. Because it has not been regularly treated, nor the area thoroughly searched for hawkweed, the infestation is now quite large and hard to eliminate—it would have been easier if it had been treated systematically earlier (C. Wickes pers. comm.).

Sometimes a weed species has been detected on an island, and perhaps even some control work initiated, but because there was no systematic follow-up surveillance and control, the species has established. For example, Atkinson (1984) first noted evergreen buckthorn on Motuhoropapa Island, in the Noises group off Auckland. He removed several plants. In 1993 the Auckland Botanical Society also removed several plants from there, and in 1994 a single plant was uprooted on another island in the group, Otata Island. The New Zealand School of Outdoor Studies were encouraged to control evergreen buckthorn when they visited these islands and they have removed 147 individuals over three visits to Motuhoropapa Island. Despite the best efforts of the various groups in the last few years, the population has expanded and is beyond easy control on the latter island at least (G. Wilson pers. comm.). Perhaps systematic action in the early 1990s would have prevented this population expansion. Fortunately current control has prevented it taking hold on Otata. A rather similar story can be told for bone-seed (*Chrysanthemoides monilifera* subsp. *monilifera*) on Red Mercury Island, first recorded in 1993 as about 50 plants, mostly so small that they could have been hand pulled. It was left to spread quietly until 1998 when the control effort had to include some abseiling to the infestations. Fortunately, further control efforts appear to have reduced the infestation to occasional plants (J. Roxburgh pers. comm.).

Mexican devil was present on the Poor Knight Islands, Northland from about the 1970s but control did not start until 1994, when several thousand plants were controlled on each visit. Eradication *may* still be possible, despite the risk of low-level re-infestation from the mainland 20 km away. Success would have been far more likely had control started sooner (G. Coulson pers. comm.).

Old man's beard (*Clematis vitalba*) has been known from Maud Island for more than 20 years. Back in the 1980s there were just a few widely separated plants that were pulled out when found (Department of Lands and Survey

1981). Since then control has been opportunistic and the old man's beard vines have probably been more difficult to spot in the increasingly dense scrub. Not until 2001 was any formal effort made to control old man's beard and now it is a huge task (M. Newfield pers. comm.). Similarly, a 1980 species list for Maud Island mentions tree mallow (*Lavatera arborea*) as present as one plant on a cliff (Department of Lands and Survey 1981). This weed, that overtops the native coastal cliff vegetation of nearby Brothers Islands, is now common around the house and main farm areas – too common for eradication to be considered (M. Newfield pers. comm.). So too, for Darwin's barberry (*Berberis darwinii*) on Stewart Island. Wilson (1982) reported it as an aggressive weed but only recently has a control programme started. Now eradication will be difficult – probably impossible – because Darwin's barberry has been used as a hedging plant on Stewart Island for years (C. Wickes pers. comm.).

Sometimes, even with the best programme of surveillance, our eradication efforts are stymied by re-invasion. For example, regular surveillance and control of boxthorn on the Sugar Loaf Islands, offshore from New Plymouth, prevents boxthorn reproducing or becoming large enough to ensnare birds. However, eradication is unobtainable because the starling roost there ensures continual re-invasion so surveillance and control must be ongoing (B. Williams pers. comm.).

DISCUSSSION

The basic principles for surveillance apply to islands as they do for the rest of New Zealand (Braithwaite 2000). In particular, islands with high conservation value or those that are vulnerable to new weed incursions are our highest priority for island weed surveillance. Taking the first point, separation from the mainland means many islands have suffered less human interference than their mainland counterparts. This physical separation and lack of disturbance makes many islands potential refuges for preservation of threatened species, thus further increasing their conservation value.

As the above examples tell, weed surveillance on islands has often been sub-optimal, allowing small weed incursions to spread and become huge infestations. In part, this is due to the several factors that increase the vulnerability of islands to new weed invasions. Visitors increase the risk of new weeds. This has been shown for mainland reserves (e.g. Macdonald *et al*. 1989) and holds true even when the effect of reserve size and species richness is taken into account (Lonsdale 1999). It has also been shown for islands, for example, on the French subantartic islands (Frenot *et al*. 2001). On these islands, the main vectors of alien species are the routine supply ships. The risk of new weed introductions increases with the number of visitors, the frequency of visits, and varies with the type of visitor. Even people doing weed control can pose a risk. They must be particularly vigilant to ensure they don't transport weed propagules from control sites to clean sites. While more visitors can mean more weeds, the converse is that visitors improve the chances of a new weed incursion being spotted early. It is anticipated that the advent of a systematic approach to surveillance, including good follow-up, will change the balance in favour of a net reduction in weed problems.

Islands that have been or are occupied, are more likely to get weeds. If gardens have been established then the risk is greater (Sullivan *et al.* 2001). Similarly, islands are more vulnerable to weed invasion if they are down-wind from a source of wind-borne seeds, or are visited by birds from areas with bird-dispersed seeds. Often it is natural disturbance such as erosion or slips that make it possible for these weed propagules to establish. Historically, boat owners have accidentally brought weeds to islands or even deliberately planted trees or other exotic plants on islands. Although many islands have restrictions on access, these can be difficult to enforce. For example, people on vessels visiting Little Barrier Island are required to obtain a permit before landing, but visitors without permits occasionally land and spend time on the island before being asked to leave (Braithwaite 2000).

Already the Plan has served to increase weed surveillance activity on islands. This is especially so for some islands in the Hauraki Gulf (Fig. 2); Great Barrier, Mokohinau, Rangitoto and Tiritiri Matangi Islands were searched in early 2001. Banksia (*Banksia integrifolia*) was discovered on Rangitoto and the bidibid (*Acaena agnipila*) and *Senna* sp. were found on Tiritiri Matangi Island (G. Wilson pers. comm.).

The efficacy of weed surveillance on islands is improved by having a search image. Weed Surveillance Lists have been prepared: lists of species to watch out for in particular geographical areas, with accompanying species information sheets and illustrations (Newfield 2001). A list comprises species that are not recorded in that area, or have very limited distribution, yet have the potential to become invasive weeds there. Often the species will be weeds in an adjacent area. The lists are not comprehensive but usually focus on species that are the most damaging or the most likely to appear. An example of a special list for islands is given in Table 1. Weed surveillance on islands is greatly enhanced by having a full plant list for each island, kept up to date by a botanist who is responsible for identifying specimens and giving advice on the status of any plant species found on the island. This situation applies to some New Zealand islands such as Raoul Island (A. Warren pers. comm.).

Despite the increase in purposeful surveillance, many reports of weeds on islands will still come, as in the past, from fortuitous finds by members of the public (Braithwaite and Timmins 2000). For example, members of the Auckland Botanical Society found evergreen buckthorn in the Noises as described above. Similarly, on Mana Island, members of Wellington Botanical Society found some brush wattle (*Paraserianthes lophantha*) and hacked them out (Timmins *et al*. 1987). Alerted to its presence, the re-

Table 1 Weed Surveillance List for Hauraki Islands, North Island, New Zealand.

These species have not been recorded, or are of very limited distribution, on islands in the Hauraki Area. They have the potential to become invasive weeds on Hauraki islands (i.e. those islands in the jurisdiction of the Department of Conservation's Hauraki Area Office).

Common name	*Scientific name*	Description
monkey apple	*Acmena smithii*	Tree. Scented shiny green leaves, white flowers, large white berries.
century plant	*Agave americana*	Huge succulent with rosette of thick, pointed leaves. Tall flower stalk with yellow flowers.
Mexican devil	*Ageratina adenophora*	Sprawling shrub, red stems, triangular leaves with sticky hairs, white flowers.
mist flower	*Ageratina riparia*	Similar to Mexican devil, but with longer, narrower leaves.
bangalow palm	*Archontophoenix cunninghamiana*	Like the native nikau palm, but with long straight crownshaft (not goblet-shaped), leaves Y-shaped when young.
moth plant	*Araujia sericifera*	Vine with stark white flowers, large green pods, and sticky milky sap.
climbing asparagus	*Asparagus scandens*	Climber, wiry stems, fine foliage, orange berries, underground tubers.
banksia	*Banksia integrifolia*	Tree. Coarse long narrow leaves. Green-yellow bottlebrush flower. Hard woody seed cone with brown felt.
Darwin's barberry	*Berberis darwinii*	Shrub. Small, dark green holly-like leaves and small orange flowers.
climbing spindleberry	*Celastrus orbiculatus*	Deciduous vine, foliage yellow in autumn, fruit with "spindle" splits when ripe, red seed inside yellow capsule.
bone-seed	*Chrysanthemoides monilifera* subsp. *monilifera*	Bushy shrub. Yellow daisy-like flowers. Orange berries with hard black seed.
pampas grass	*Cortaderia selloana*	Large clump-forming grass to 3 m tall. Fluffy seed heads on tall straight stems.
cotoneaster	*Cotoneaster* spp.	Spreading evergreen shrub, smooth leaves, small scarlet berries.
fairy crassula	*Crassula multicava*	Creeping succulent. Small, red-spotted leaves. Delicate pink flowers, plantlets.
needlebush	*Hakea sericea*	Large spreading shrub, leaves spiny, beaked woody fruit.
lantana	*Lantana camara*	Scented shrub covered in small prickles, with mixed-coloured flowers: cream and pink, yellow and dark orange.
boxthorn	*Lycium ferocissimum*	Dense, very spiny evergreen shrub, orange fruit.
drooping prickly pear	*Opuntia vulgaris*	Huge cactus, flat oval stems, long spines, yellow flowers, brownish-red fruit.
saltwater paspalum	*Paspalum vaginatum*	Stoloniferous grass growing in intertidal areas.
black passionfruit	*Passiflora edulis*	Vine with white passion-flower, large black fruit.
banana passionfruit	*Passiflora mollisima, P. mixta*	Vines with pink passion-flower, long yellow fruit.
Phoenix palm	*Phoenix canariensis*	Large palm with spines and massive basal stem.
cherry or plum	*Prunus* spp.	Deciduous small trees, crimson drupes.
evergreen buckthorn	*Rhamnus alaternus*	Thorny shrub with glossy, toothed leaves, green flowers, berry black with three seeds.
woolly nightshade	*Solanum mauritianum*	Smelly shrub, large soft leaves, purple flowers, clusters of large yellow berries.
brush cherry	*Syzygium australe*	Tree, scented shiny red-tipped leaves, white flowers and oval pink berries.
windmill palm	*Trachycarpus fortunei*	Tall palm, matted fibres cover stem, fan-shaped leaves, bluish-black berries.

These species are not the only possible new weeds. If you find any of the above plant species, or a plant that is unfamiliar or seems out of place, contact weed staff at the Department of Conservation Hauraki Area Office or Environment Waikato. Summary sheets with information and an illustration are available for each species.

List supplied by David Stephens, Waikato Conservancy, Department of Conservation, Hamilton.

serve managers removed the other enclaves of brush wattle found over successive years, so now Mana is free of adult plants of this weed (J. Sawyer unpub. data). This stream of casual weed reports will be enhanced by increasing public awareness of weeds through talks, newspaper items and signs, and from encouraging groups such as botanical societies to report their fortuitous finds (Timmins and Blood in press).

CONCLUSION

Weed surveillance favours the early detection and control of weeds: it improves our chances of eradication and minimises ecological damage. This weed wisdom is imperative on islands with high conservation values, difficult access and infrequent visitors. The alternative to active surveillance is to wait until an infestation is found by chance; a risky approach on islands with high conservation values and/or vulnerability to weed invasion. New Zealand's track record for weed surveillance on islands is variable, but it is anticipated that chances of success will improve with the recent development of the Department of Conservation's Weed Surveillance Plan. Our chances of finding new weed incursions increase by searching in vulnerable places such as track edges, boat ramps, slip faces, bush margins, coastal fringes and under starling roosts. Weed surveillance on islands is greatly enhanced by having a list of weed species to watch out for on each island. We want all visitors to islands to be alert for new weeds – those on the list as well as the unexpected – and to report any plant that seems out of place.

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Eradication of rabbits and mice from subantarctic Enderby and Rose Islands

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Abstract In 1993 rabbits (*Oryctolagus cuniculus cuniculus*) were eradicated from Enderby (700ha) and Rose (80ha) islands in the New Zealand subantarctic Auckland Island group. This was achieved by a widespread poison campaign followed by an intensive second phase which included hunting with a dog, spotlighting and trapping. During the poison campaign a helicopter was used to apply a cereal pelleted bait incorporating the anticoagulant toxin brodifacoum to both islands. Mice (*Mus musculus*), which were present on Enderby, disappeared during the poison campaign and appear to have been eradicated during this phase. The potential impacts to non-target species were assessed prior to the operation. Although the poisoning had a notable short-term impact on skua (*Stercorarius skua lonnburgi*) numbers there has been no obvious long-term impact on any non-target species. Rabbits and mice were the last of several introduced mammal species to be removed from Enderby and Rose. Without them the unique ecological values of these islands have a chance to recover.

Keywords Eradication; rabbits, *Oryctolagus cuniculus cuniculus*; mice, *Mus musculus*; Auckland Islands; Enderby Island.

INTRODUCTION

The Auckland Islands are an uninhabited subantarctic group lying 460 km south of New Zealand, at approximately 50°40’ S, 166°08’ E. The group consists of two large and five smaller islands and numerous small islets (Fig. 1). The two largest are Auckland Island (46,000 ha), which rises to a maximum altitude of 664 m, and Adams Island (9900 ha), which rises to 667 m. At the northern end of the group, around Port Ross, there are four smaller, low-lying islands, the largest two of which are Enderby (700 ha) and Rose (80 ha) Islands (Fig. 1). The Auckland Islands are gazetted as a National Nature Reserve (Reserves Act 1977) and are of international ecological importance because of their particularly diverse and unique biological communities, which include many endemic species of plants and animals such as Auckland Island teal (*Anus aucklandica aucklandica*), snipe *(Coenocorypha aucklandica*) and Auckland Island rail (*Rallus pectoralis aucklandica*) (Penniket *et al.* 1987). They are also an important breeding ground for marine mammals and seabirds including New Zealand sea lion (*Phocarctos hookeri*) and wandering albatross (*Diomedea exulans*).

Since their discovery in 1806, the islands have been subjected to significant human impacts. These include the introduction of a variety of alien mammals, at first by sealers or as food for castaways, and later during attempts at farming. Some of these animals have died out naturally, but in recent times Auckland Island has continued to be host to pigs (*Sus scrofa*), goats (*Capra hircus*), cats (*Felis catus*), and mice; Enderby Island to cattle (*Bos taurus*), rabbits, and mice; and Rose Island to rabbits (Fig. 2) (Taylor 1968, 1971).

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Since 1987, New Zealand’s Department of Conservation has actively pursued a policy, set out in the Management Plan for these islands, to eradicate all alien animals as soon as is feasible (Penniket *et al.* 1987). Goats were eradicated from Auckland Island between 1989 and 1991 (A. Cox pers. comm.). The majority of cattle were removed from Enderby in 1991, with eradication being completed by the rabbit eradication team in 1993. In 1991, the feasibility of removing rabbits from Enderby and Rose Islands was investigated. Based on the results of that investigation, a programme aimed at the total removal of rabbits

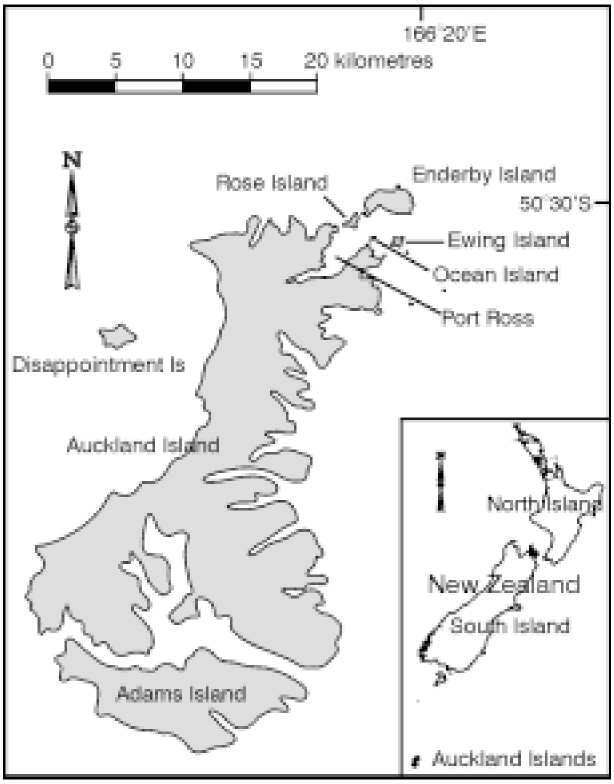

Fig. 1 The Auckland Islands showing the location of Enderby and Rose Islands, and other places mentioned in the text.

Pages 319-328 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

from both islands was instigated. During the investigation and planning for this operation it became apparent that there could be an opportunity to eradicate mice from Enderby at the same time as rabbits. This was adopted as a secondary aim of the eradication programme.

The proposed programme presented some challenging features, including:

- the difficulty of carrying out an eradication operation in an isolated situation far (460 km) from the New Zealand mainland, under unfavourable climatic conditions;
- the size of Enderby, which at 700 ha, is much larger than any island from which rabbits and mice had previously been eradicated (Round Island at 151 ha and Mana Island at 217 ha respectively) (Merton 1987; Hook and Todd 1992).
- the presence of some indigenous birds, considered to be at risk from some of the methods used in the eradication programme.

This paper reports the progress of the eradication programme from initial bait trials in 1991. It covers the 1993 eradication operation and the following period of monitoring.

Description of Enderby and Rose Islands

Both islands are comparatively low-lying, with Enderby Island rising to a maximum altitude of about 45 m and Rose slightly higher at 48 m. Except for two sandy beaches on Enderby and a few sections of steeply sloping shoreline, both islands are almost completely surrounded by coastal cliffs. These rise to over 30 m on their northern and western shores, but are generally lower on the southern and eastern coasts. Apart from about 30 ha of sand dunes on Enderby Island, both islands are covered in a thick blanket of peat, which in many areas is waterlogged.

Since their discovery, modification to the original vegetation of these two islands has been quite severe, mainly through the use of fire during failed farming attempts in the late 1800s and the presence of introduced mammals (Fig. 2) (Taylor 1968, 1971).

Along the southern and eastern sides of both islands, and covering about one quarter of their surface areas, is a belt of southern rata (*Metrosideros umbellata*) forest and scrubland. The rest of Rose Island is predominantly covered by large areas of *Poa litorosa* tussock grassland interspersed with areas of short sward vegetation. On Enderby, the centre of the island and about one third of its area is moorland dominated by the shrub *Cassinia vauvilliersii* and cushion plant *Oreobolus pectinatus* interspersed with isolated patches of rata forest and *Myrsine divaricata* scrub. Around the coast is a band of short sward vegetation, which is extensive on the northern and western side, and up to several hundred metres wide in some places.

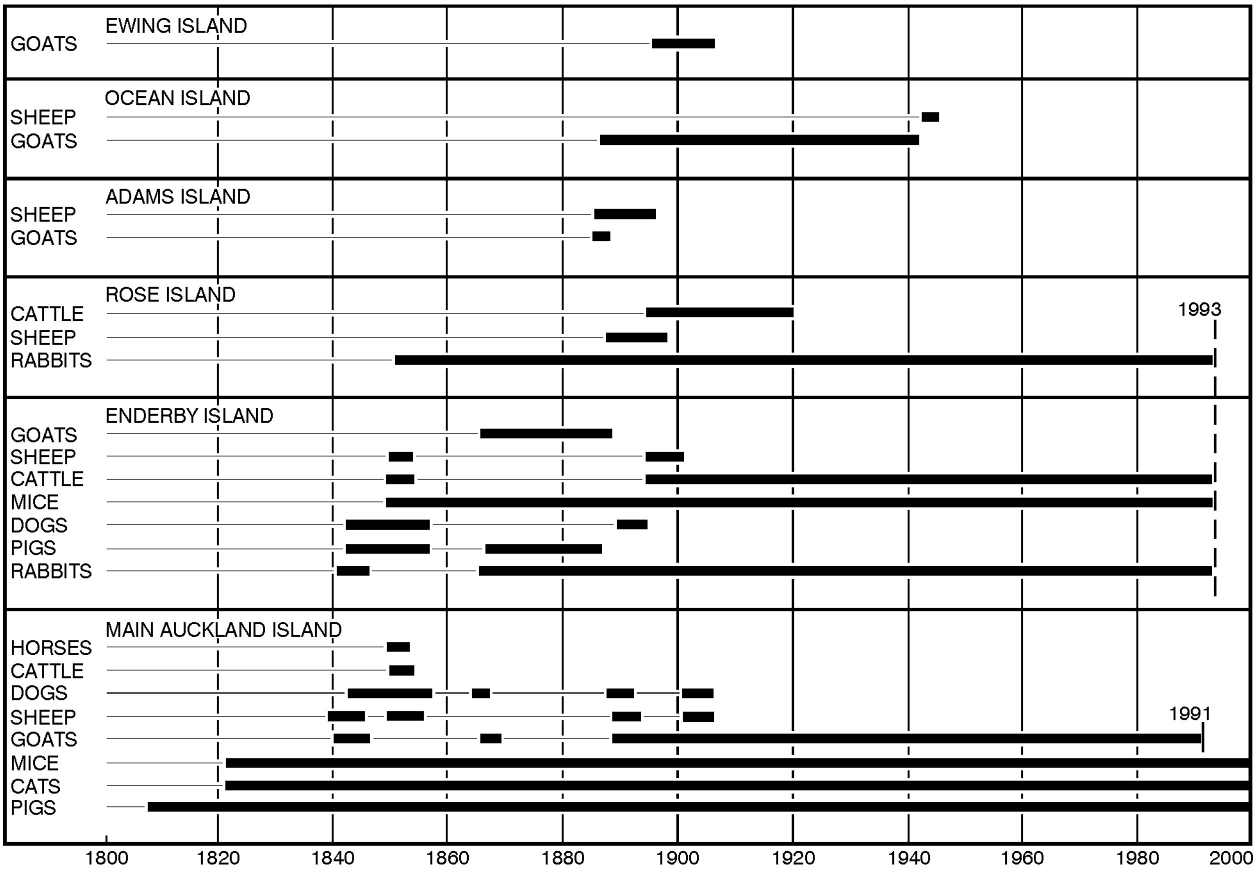

Fig. 2 Historical summary of introduced mammals on the Auckland Islands (updated from Taylor 1968). The thick bars indicate when each species was present.

Prior to the eradication, cattle and rabbit browsing had confined the large-leafed herbaceous plants *Stilbocarpa polaris* and *Anisotome latifolia* to cliff areas and *Pleurophyllum criniferum* had been eliminated from both islands. On Enderby Island, tussock grasslands had been almost eliminated and replaced with a closely-cropped sward made up of a mixture of both introduced and native plants.

History and effects of rabbits on Enderby and Rose Islands

Rabbits were liberated on Rose Island in about 1850, during a period of attempted settlement in the Port Ross area. The liberation was probably of mixed stock, or there may have been other unrecorded liberations, as the population there included agouti and silver black rabbits, both with and without white markings (Taylor 1971).

The rabbits on Enderby were descended from 12 animals liberated in 1865 to establish a population as food for castaway mariners (Taylor 1971). They came from the Acclimatisation Society of Victoria in Australia and belonged to the French breed known as "Argenté de Champagne" (or French Blue). The rabbits on Enderby had bred true to this type (Taylor 1971). Argenté de Champagne is now a fairly rare breed and it was thought that the Enderby population might have been the last true wild population left in the world (B. W. Glentworth pers. comm.). Before the eradication, 49 rabbits were recovered from Enderby Island by the Rare Breeds Conservation Society of New Zealand and the Department of Conservation, and brought back to New Zealand to form the nucleus of a managed captive population.

During a visit to Enderby Island in 1991, Glentworth mapped the density and distribution of rabbits by walking several circuits of the island during periods of high rabbit activity at dawn and dusk and at night with a spotlight (Fig. 3). He then compared the number of rabbits seen as well as the amount of scratchings and droppings observed to areas of known rabbit density on mainland New Zealand. From this method he estimated the total population at 5000-6000 animals. On Rose Island, density, distribution, and numbers of rabbits were much harder to estimate because of the thick ground cover of *Poa litorosa*, but the total population was probably 300-400 individuals. Estimated rabbit density for both islands ranged from >35 animals per hectare in the more favoured habitat around Sandy Bay on Enderby Island, to <2 animals per hectare in less favourable habitat. In general, rabbits were more numerous on Enderby than Rose Island. Rabbit sign was found over all areas of both islands, including the rata forest (B. W. Glentworth pers. comm.).

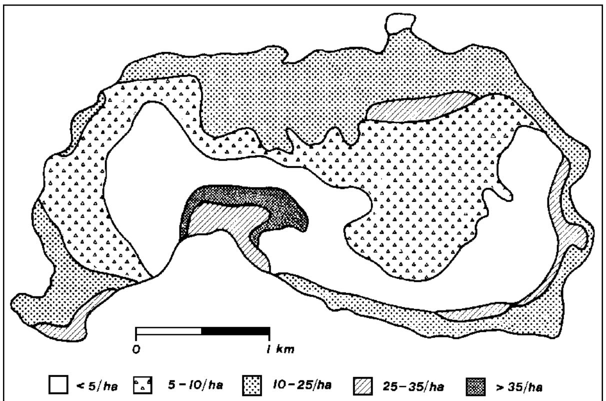

Fig. 3 Distribution and estimated density of rabbits on Enderby Island, Feb 1991 (Glentworth pers. comm.)

In addition to their impacts on vegetation, rabbits also reduced the survival of New Zealand sea lion pups on Enderby Island. New Zealand sea lions are found only within New Zealand waters and are a threatened species. Sandy Bay on Enderby Island is an important breeding site for them. The numerous rabbit burrows around Sandy Bay proved a hazard to sea lion pups, which became trapped in them and died. Mortality from this cause was estimated as up to 10% of pups on Enderby Island in some years (Penniket *et al.* 1987).

History and effects of mice on Enderby Island

Mice were accidentally introduced to Auckland Island during the main period of sealing activity in the early 1820s (Taylor 1968). They probably arrived on Enderby about 1850, when there was an increase of human activity on the island associated with a period of attempted settlement in the Port Ross area (Taylor 1971). They have thrived since then and, prior to the eradication programme, were widespread across the whole of the island and were often seen in large numbers around the hut and old boatshed at Sandy Bay.

Little is known about the direct effect mice have had on Enderby Island ecosystems, but it is likely that predation had a profound influence on the invertebrate fauna. It is also possible that they have had some influence on vegetation through eating seeds and young shoots.

METHODS

Eradication strategy

The rabbit eradication programme was designed to follow a similar strategy to that used for the successful eradication of rabbits from Round Island, Mauritius, in 1986 (Merton 1987) and involved two distinct phases. The aim of the first phase was to lower the total rabbit population quickly and substantially. This would be achieved with two aerial applications of poison bait spaced approximately 14 days apart and applied to the total surface area of both islands.

Experience gained from other rabbit poisoning operations indicated that it is unrealistic to expect all rabbits exposed to poison bait to succumb. After the poison operation on

Round Island, 14 rabbits were shot during the following period of hunting. At least one of these showed no signs of poisoning and had apparently not fed on bait (Merton 1987). During the eradication of rabbits from Whale Island in the Bay of Plenty, New Zealand, some rabbits avoided eating some types of poison bait, despite being exposed to several applications (Jansen 1993). Thus a second phase was planned to track down and destroy any rabbits left after the initial poison operation.

The second phase was to start one week after the second application of poison, by which time most rabbits affected by the poison were expected to have died. A variety of methods were chosen including hunting with a dog, shooting, spotlighting, and trapping. This would require a team of dedicated and experienced hunters with a dog to be on the islands for two months after the poison operation. Visits to the islands by the same or a similar team were also planned for the following years until no live rabbits remained.

Although mice had been eradicated from five islands (up to 217 ha) off the New Zealand coast, using ground-administered second-generation anti-coagulant poisons (Veitch 1994; Hook and Todd 1992; Brown 1993), they had never been eradicated using only aerial application of a poison bait. However, recent success at eradicating Pacific rats (*Rattus exulans*) from islands up to 225ha (Veitch 1994) using a single aerial application of poison bait similar to the type we intended using for rabbits, led the project planning team to believe there was a strong possibility that mice could be eradicated at the same time as rabbits. Although the major effort of this campaign was always focused towards rabbits, some steps, as noted later in this paper, were taken with the application of poison to increase the chances of also eradicating mice.

To establish whether eradication had been successful required the island to be monitored for two years after the last sign of live rabbits and at least four years after the last mouse sign was seen. If rabbit sign was still found three years after the initiation of the eradication programme, progress would be reviewed with serious consideration given to continuing the programme.

Logistics for the Eradication Programme

A team of four people and one specially trained rabbit tracking dog were stationed on Enderby Island from 9 February to 8 May 1993, and two of the original four people and the same dog were stationed there from 20 January to 19 February 1994. Visits to Rose Island were made regularly through these periods using a 3.5 m dinghy with outboard motor. A helicopter flew from New Zealand to the Auckland Islands twice during the first period to spread poison bait.

Choice of bait

Three important aspects were considered when deciding the most suitable bait: (1) palatability to target and non-target organisms, (2) ease of handling and storage, (3) effectiveness in wet conditions. Two bait types commonly used for rabbit poison operations in New Zealand – diced carrot and cooked oats – were not considered because of the logistical problems of keeping bait fresh, the need to prepare bait on site, and potential non-target problems. Instead, trials concentrated on several types of manufactured cereal-based pellets. These were considered the most appropriate bait type for this operation because: (1) they may be pre-loaded with toxin and kept in storage for extended periods of time, (2) no preparation is required on site, and (3) they are light and easy to handle. They are also known to be highly palatable to rabbits and rodents.

An important consideration was how well bait would stand up to climatic conditions and remain effective once laid. The Auckland Islands have a relatively wet climate. An average of over 1400 mm of rain falling over more than 300 rain days per year was recorded in the Port Ross area during 1941-1945, the only period continual records have been kept (De Lisle 1965). Cereal pellets are known to break down fairly quickly in wet conditions. The possibility of toxin being leached out of bait was also considered. For each poison drop to be most effective, bait was required to remain palatable and toxic for at least three nights after the drop.

During two trips to Enderby Island in 1991 and 1992, acceptance and preference trials of several bait types were conducted *in situ* (B. W. Glentworth pers. comm.; W. P. Costello pers. comm.). The bait preference trials indicated that Enderby Island rabbits ate all bait types tested in preference to natural foods and had no significant preference for any one bait type. However, it was noted that "Wanganui No. 7" (manufactured by Animal Control Products, Wanganui, New Zealand), while by no means waterproof, had superior weathering characteristics to the other bait types tested. For this reason Wanganui No. 7 was chosen for this operation.

A bait acceptance trial with Wanganui No 7 was also conducted using "Rhodamine B" (Tetra-ethyl rhodamine) as a biotracer. Baits were presented over a 5 ha area of high rabbit density in a manner resembling as closely as possible the planned bait drop. After three nights a sample of 46 rabbits was shot. All tested positive, indicating total acceptance of the bait by rabbits (W. P. Costello pers. comm.).

During the bait trials mice showed considerable interest in all bait types tested. In some trials up to 15% of the bait was consumed or partially eaten by mice (B. W. Glentworth pers. comm.).

Choice of toxin

Brodifacoum was chosen as the most appropriate poison for this campaign. A second-generation anticoagulant, it was the poison used to eradicate rabbits from Round Island (Merton 1987) and has also been used for the majority of successful rodent eradications from islands around New Zealand. Advantages over most other available poisons included:

- It is extremely toxic to rabbits and mice. The LD50s for these species are 0.29mg/kg and 0.4mg/kg respectively (Hone and Mulligan 1982). This allows very low loadings of toxin, which reduces the risk of primary and secondary poisoning of non-target animals.
- The onset of symptoms does not occur for several days after poison has been consumed. Therefore, animals have plenty of time to consume a lethal dose before feeling any ill effects and poison shyness is unlikely to develop. This means that pre-baiting is unnecessary.
- A lethal dose of poison can be ingested in a single feed or accumulated during several feeds over an extended period of days. This makes it more likely that every animal will obtain a lethal dose of poison, thereby reducing the chances of sub-lethal poisoning and subsequent development of poison shyness.
- Animals normally take several days to die after ingesting a lethal dose of poison, becoming progressively weaker in the hours before death. Because of this, many animals die underground or in cover, therefore reducing the risk of secondary poisoning of scavenging birds such as skuas.
- Brodifacoum is not soluble in water and is therefore slow to leach from baits in damp conditions. When released it binds onto organic matter in the soil and is rendered inert. Soil microorganisms then slowly degrade it over a period of 3-6 months (Shirer 1992).
- Brodifacoum is relatively safe from an operator's point of view. Vitamin K1 is an effective antidote.

For this operation, brodifacoum was mixed into the bait at manufacture, at a concentration of 20 parts per million (0.002%).

Managing the threat to non-target species

An important aspect of the eradication strategy was to evaluate and manage as far as possible the risk to non-target species. Experience from previous poison campaigns in New Zealand indicated that there was no significant risk to most species present on Enderby and Rose Islands. However, subantarctic skua (*Stercorarius skua lonnburgi*) and Auckland Island teal (*Anas aucklandica aucklandica*) were cause for some concern. Both species were considered potentially at risk from eating poison baits. Skuas, known scavengers and hunters of rabbits on Enderby and Rose Islands, could be at further risk of secondary poisoning from eating poisoned rabbit carcasses.

All main islands of the Auckland Island group have breeding populations of skuas. Seven pairs on Enderby and three on Rose were recorded breeding in 1991 and 1993. Past observations from other islands indicated that this was less than half the total Auckland Island breeding population. Teal exist on all the large offshore islands in the Auckland Island group. Williams (1986) estimated that the total population was "at least 500 birds". Of these he estimated approximately 50 lived on Rose and 76 on Enderby Island.

As a simple assessment of risk, bait acceptance trials were conducted. Teal in captivity and in the wild on the Auckland Islands were exposed to a non-toxic version of the baits to be used for the poison operation. In both cases the birds showed little interest. Skuas on Enderby Island were also exposed to non-toxic baits. They showed little interest in them, except for two occasions when two birds were seen to eat a few pellets.

Although this was a positive outcome it was not considered conclusive and the decision to undertake the poison operation was made acknowledging there was some risk that was difficult to accurately quantify. This was considered acceptable because the poison operation was to be a one-off event and only a portion of the total Auckland Island population of both species would be exposed. If, in the worst possible scenario, the Enderby and Rose Island populations were completely lost, the islands could be recolonised by birds from other islands in the group.

All bait used during the poison operation was dyed green. This is known to reduce its attractiveness to birds (Caithness and Williams 1971).

Seasonal timing of the poison drop

Two major factors influenced the timing of the poison operation. It was important that it be done well outside the time when rabbits were breeding, when many young rabbits live underground and are not vulnerable to poison. From ageing data of a sample of rabbits shot on Enderby Island in 1991, Glentworth (pers. comm.) concluded that the breeding season extended from July through to December with a peak around September/October. Observations from other visits to the island indicated that breeding sometimes carried on into January. The second major factor was weather. Because cereal pelleted bait breaks down fairly rapidly in wet conditions, it was desirable to spread it in the driest time of the year. Taking these factors into account, mid February was chosen as the most appropriate time for the main poison operation.

Bait sowing method

Two applications of bait were made to both islands. This ensured complete coverage, minimising the chance of gaps. It also helped ensure that enough bait was laid in areas of very high animal density and that bait remained available long enough for every rabbit and mouse to be able to consume a lethal dose of poison. Having more than one drop also reduced the risk of failure due to rain. As further insurance, enough resources were on hand for a third poison drop.

Because of the size of the area to be treated and the need to sow bait quickly when weather conditions were suitable, helicopter application was considered the only practical method. Bait was spread using an AS 350 B "Squirrel" helicopter with an under-slung spreader. The spreader was purpose built for this type of operation and had an adjustable aperture which allowed the rate at which bait was being sown to be regulated. It was fitted with a rotary spinner, which threw bait out to 20 m either side of the spreader. This gave a 40 m wide strip (or swath) of bait coverage for each pass of the helicopter. Parallel flight lines were flown north to south across the islands 35 m apart to ensure there was an overlap between swaths. To help with this on Enderby Island, one person walked along a line running east to west across the island marking 35 m intervals measured with a hip-chain. This gave the helicopter pilot a reference point for each pass. Using this method, accurate coverage was achieved during both poison drops. On the much smaller Rose Island this was not necessary, as it was easier to keep track of where bait had been spread.

Phase one – the poison drop

The services of a professional weather forecaster in New Zealand were used to help choose a relatively dry and settled period of weather for the bait applications. We made contact via radio and were able to receive forecasts for the Auckland Islands area when needed.

The first application of bait was sown on 15 February 1993, and the second, after having been delayed slightly by weather, was 18 days later on 5 March. The applications were spaced so that most rabbits poisoned from the first had died before the second was applied.

Bait was applied at 5 kg/ha over the whole of Rose Island on both drops. On Enderby Island the rate was the same, with the exception of 100 ha of heavily rabbit-infested country. This area was treated at 10 kg/ha during the first drop. On the second drop only 20 ha were treated at this increased rate.

During both applications on Enderby Island, special care was taken to ensure bait fell on all areas where mice might live. These included small ledges on cliff faces and small beaches and rock platforms at the base of coastal cliffs, which wouldn't have been treated in a campaign against rabbits only. Bait was also sown at a rate high enough for confidence that all mice would have access to a lethal dose of poison.

As a check that bait was being sown at the correct nominated rate, Enderby Island was divided into 100 ha blocks. As each block was completed, the quantity of bait sown in that block was checked.

Less than 1 mm of rain was recorded at Enderby Island for the 10 days after the first bait drop and only 14 mm fell in the seven days after the second. This was not considered enough to reduce the effectiveness of the bait in any way.

Phase two – the follow up

From one week after the second drop until leaving Enderby Island eight weeks later on 8 May, and during the second five week visit to the islands one year later in Jan/Feb 1994, the field team (assisted by a trained rabbit-tracking dog) concentrated on locating and destroying any rabbits remaining after the poison operation. The dog was the most effective method used in this part of the operation. The dog found and followed rabbit scent, generally flushing rabbits from cover when they would either be shot or chased down burrows from where they were dug up and destroyed. Occasionally, the dog caught a rabbit before it could get to a burrow. This was especially so for young animals. On some occasions the dog would consistently follow rabbit scent in a particular area but we would not be able to find the rabbit and would need to visit the area repeatedly before accounting for that individual. In this way some rabbits were hunted over a period of up to 10 days before finally being destroyed. By this process we became familiar with their movement patterns and found that these rabbits ranged within discrete areas, which in some cases could be up to 25 ha.

The dog also gave us a high degree of confidence in establishing the absence of rabbits. For example, once we had thoroughly searched an area a number of times with the dog and found no sign we could be reasonably confident that there no were rabbits present.

Spotlighting was another method used to find and destroy rabbits. Teams armed with a shotgun and spotlight (operated from a 12 volt battery) would hunt on calm, dry nights when conditions were considered most favourable for rabbit activity, or if a known rabbit had not been caught during day hunting expeditions. Although this was not nearly as productive as hunting with the dog, some rabbits were destroyed using this method.

Traps were used on only one occasion. The last rabbit on Rose Island proved very difficult to catch using the methods outlined above. The dog could consistently find and track its scent but because of heavy cover in the area where the rabbit was living it proved impossible to shoot or chase to ground. Spotlighting also proved ineffective. As a last resort six "Lanes Ace" leg-hold traps were set in the area it was known to be using. This rabbit was caught on the second night the traps were set.

In addition to these methods many hundreds of hours were spent systematically and meticulously searching both islands for rabbit sign. Once sign had been located we never failed to eventually find and kill the rabbit that had left it.

During this phase the team kept a careful watch for any sign of mice surviving on Enderby Island.

RESULTS

Impacts on rabbits and mice

On Enderby, the first dead rabbit was found four days after the initial poison drop. From then on the rate of mortality steadily climbed, to peak at around 10 days after the drop. Rabbits continued to die from the effects of the first drop right up to the point where the second drop was sown. At this point, from casual observation, mortality appeared to have reached in excess of 90%. After the second application of poison, most of the remaining rabbits quickly disappeared and by mid March live rabbits were extremely rare (Fig. 4). Rose Island was not monitored as closely as Enderby before and during poisoning, but it seems likely from observations made during regular visits that the kill rate there closely resembled that on Enderby.

From mid March to early May 1993, 22 live rabbits were found and killed on Enderby and 12 were found and killed on Rose. Some of these animals showed obvious symptoms of poisoning and would probably have died, given time. However, approximately 70% of the survivors (25 animals), showed no obvious sign of having taken poison. In each of these cases we know that the animal had access to bait because we had either checked the area where it was living for bait coverage soon after the poison drop or there was still sign of bait present when the animal was found. Without having samples from these rabbits analysed for traces of brodifacoum it is impossible to be certain, but it seems likely that at least some of these animals avoided eating the bait.

The last rabbit was caught on Enderby on 12 April, and on Rose on 27 April 1993. This gave a further four weeks on Enderby and two weeks on Rose for careful searching, during which time there was no sign of live rabbits. During a visit to the islands specifically to search for rabbit sign in Jan-Feb 1994, no indications of rabbit presence were found. Following this, a careful search of both islands by party members of expeditions stationed on Enderby over the summer of 1994-1995 also failed to find any sign of rabbits. It would appear that rabbits were eradicated during the 1993 expedition.

Several mice showing obvious signs of poisoning were found within three days of the first application of bait on Enderby Island. From then on, all sign of live mice on the island quickly disappeared, and the remains of dead mice were commonly seen during searches of the island over the remainder of the 1993 trip. No mouse sign has been seen on the island since then despite a careful search for sign during the 1994 trip and on all subsequent visits to the island. It appears that mice have been eradicated from Enderby Island.

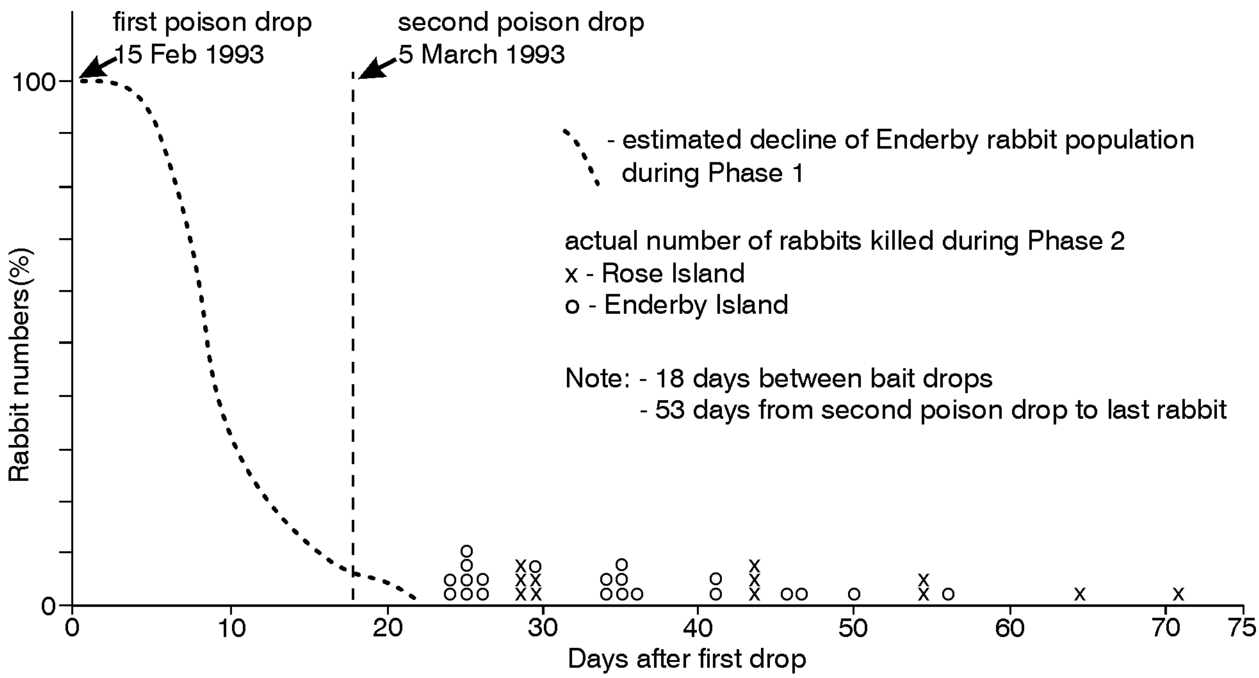

Fig. 4 Estimated decline of rabbits during Phase 1 of the operation on Enderby Island (dotted line) and the number of rabbits killed during Phase 2 for Rose (x) and Enderby (o) Islands shown with one symbol for each animal killed.

Impact on non-target species

Four Auckland Island teal carcasses were found on Enderby and three on Rose Island in the two months following the poison operation. Although there is no direct evidence that these birds died from eating poison, it seems likely that at least some of them did. Despite these deaths, live teal were encountered frequently on both islands after the poison operation in 1993 and during the 1994 visit. From casual observation there appeared to be little difference in frequency of encounters before and after the poison operation. It therefore appears that although some teal were poisoned, the total lost was relatively few.

Approximately two thirds (40 birds) of the Enderby and Rose Islands skua population died during the poison operation, apparently from eating poisoned bait. We knew skuas had eaten poison because the green dye used in the bait was visible in their droppings. Although the possibility of a loss had been predicted, the fact that the majority of birds were poisoned from eating bait, rather than from secondary poisoning, was unexpected. About 20 skuas were still seen around both islands by the time we departed at the end of the 1993 trip. During the 1994 trip the numbers seen were about the same or slightly greater though only one pair was found breeding on each island. Since 1994 the skua population on both islands has recovered to near pre-eradication levels. Fifty-two birds including five breeding pairs were counted on Enderby in the summer of 2000-2001 and two pairs were found breeding on Rose.

In addition to teal and skua deaths, about 10 blackbirds (*Turdus merula*) were also found dead or showing obvious signs of poisoning following the poison operation. Blackbirds were still seen frequently later in the trip and in the following year. The impact on the local population was apparently not great. Although the risk to blackbirds was identified during planning for the poison operation, it was not a serious concern because they are a species introduced to the New Zealand region, common in all parts of New Zealand.

DISCUSSION

The results of the initial poison campaign were remarkably successful. Of the estimated 5000-6000 rabbits present on Enderby Island, over 99% were destroyed during the first phase of the eradication programme. Nevertheless, a significant number of rabbits apparently avoided eating bait and were destroyed during the follow-up phase. This also occurred during the eradication programmes on Round and Whale Islands (Merton 1987; Jansen 1993). This bait avoidance is particularly interesting in the case of the Enderby and Rose Islands rabbit populations, which have existed in almost complete isolation from human contact for over 100 years. Bait avoidance has clearly not developed through any direct experience of poison or baiting regimes but rather may be an innate neophobia that occurs naturally in some rabbit populations. This has implications for any similar rabbit eradication programmes and emphasises the importance of having an adequate follow-up phase built into the programme to account for the last few animals.

The dog played a key role during the follow-up phase on Enderby and Rose Islands, and was by far the most effective method used to find and destroy rabbits. Without the dog the programme would certainly not have reached its successful conclusion so quickly. Another factor contributing to the success of the second phase and the overall goal of eradication was the experience and dedication of the team on the island. Without these two factors, the programme would probably have had to continue for at least one more year and may never have reached its ultimate goal of eradicating rabbits.

The absence of any signs of mice on Enderby Island since late February 1993 suggests that they were eradicated by the poison campaign during the first phase of this programme. The Department of Conservation (DOC) has remained cautious about this outcome until now because mice can be very difficult to detect at low densities, especially on an island the size of Enderby. However, scientific parties have worked on the island and lived in the established camp at Sandy Bay where mice were common pre-eradication, for periods in excess of six weeks every summer since the eradication. They have failed to find any sign of mice at the camp or on any other part of the island. The chance of mice existing on the island and escaping detection for so long now seems to be very slight indeed.

One of the often-difficult aspects of planning and running eradication programmes is predicting and managing the impact on non-target species. In almost all cases where traps or poison are used there will be some cost to non-target species. This needs to be balanced against the overall benefits gained from pest eradication. Within New Zealand there have been over 110 successful eradications of introduced animals from islands (Veitch 1994). In almost every instance the original biota of the island and their natural ecosystems have benefited, often spectacularly so. This includes many non-target species that initially suffered during the eradication programme.

As a general rule, so long as the impacts on non-target species are not irreversible, the benefit to the island in the long term will far outweigh any short-term losses. Bearing this in mind, care needs to be taken that any measures introduced to reduce the impact on non-target species do not compromise the chances of successful eradication of the target species. These principles were applied during the planning for this eradication, and in the case of skua and teal we were prepared to sustain greater losses than actually occurred.

Although the possibility of some skua losses had been predicted, the fact that so many birds died from eating bait was unexpected. Trials had indicated skuas showed little interest in the type of bait used for this operation. However, over half the skuas on both islands were observed to have eaten bait within 10 days of the first poison application. There were two main differences between bait trials and the poison operation. During the bait trials baits were either un-dyed or were dyed red with Rhodamine B. During the poison operation baits were dyed green. The other difference was that during the poison operation baits were available over a much wider area and for a longer period than during the bait trials. Perhaps, during this time some birds learned that baits were palatable and this behaviour was passed through the population. Most birds ignored baits until they had been available for at least several days and some birds appeared to never eat baits.

Some changes on Enderby and Rose Islands were obvious almost immediately following the removal of rabbits. Many palatable plants that had continually suffered from browsing pressure are now showing spectacular signs of recovery. The predominant tussock, *Poa litorosa*, which had been severely restricted in distribution on Enderby, is now advancing quickly and invading many areas of the herbaceous sward. The megaherbs *Stilbocarpa polaris* and *Anisotome latifolia* which had previously been restricted to cliff areas are found much more widely over both islands and scattered plants of *Pleurophyllum criniferum* (not recorded on Enderby Island for many years) can now be found.

Several mammal and bird species have already, or are likely to, benefit from the absence of rabbits. The death of New Zealand sea lion pups in rabbit burrows on Enderby Island has become much less of a problem as the disused burrows collapse or are filled in. Species like Auckland Island teal and Auckland Island snipe (*Ceonocorypha aucklandica*) that are vulnerable to avian predators (such as skua and falcon) are likely to benefit as more vegetative cover and resulting habitat becomes available to them. This is especially so for Enderby, where Williams' (1986) estimate of 76 teal on the 700 ha island is compared with his estimate of at least 130 birds on neighbouring 54 ha Ewing Island, where the vegetation cover is much more intact. Yellow-eyed penguins (*Megadyptes antipodes*) which nest on both islands, may benefit as the quality of nesting cover improves. Auckland Island rail (*Rallus pectoralis*), a rare Auckland Island endemic, very vulnerable to introduced and avian predators and presently known only from Adams and Disappointment Islands, may in time colonise or could be introduced to these islands as habitat improves.

Although it is accepted that modified habitats may not return to their pristine condition after introduced animals have been removed, it is expected that these two islands will reach a condition closely resembling it. This could take longer than 50-100 years and may require the careful management of some weed species in the interim.

Despite having to operate in a very remote and sometimes difficult environment, the eradication programme was very successful and ran smoothly. There were a number of important factors that contributed to this:

- everyone involved had a single clear objective: eradication of rabbits;
- total dedication to the objective by all staff involved and by the Department of Conservation in general;
- careful planning which acknowledged that once on the island the team would have to operate in almost complete isolation from mainland support. This meant planning for many different eventualities and having flexibility in the programme and the resources on hand to deal with them;
- good technology, including a very efficient and potent toxin, pre-manufactured bait which was easy to handle, durable, and highly palatable to the target species, and the use of helicopters which made the delivery of bait to large and inaccessible areas fast and accurate;
- adequate resources, including funding and the ability to bring in suitably skilled and experienced staff.

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Interactions between geckos, honeydew scale insects and host plants revealed on islands in northern New Zealand, following eradication of introduced rats and rabbits

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Abstract Invasive species that reach islands can have effects that ripple through communities. As a corollary, once invasive species are removed, the responses by resident species may also have ripple effects, sometimes with outcomes that are unpredicted. One such unpredicted response is reported on islands off north-eastern New Zealand following the removal of rabbits (*Oryctolagus cuniculus*) and Pacific rats or kiore (*Rattus exulans*). As composition of the vegetation changed and geckos became increasingly abundant, a source of energy for the geckos was revealed: honeydew produced by the scale insect *Coelostomidia zealandica* (Hemiptera: Margarodidae) infesting ngaio (*Myoporum laetum*) and karo (*Pittosporum crassifolium*). Honeydew may have significant effects on the carrying capacity of invertebrates and birds in mainland forests of New Zealand. However, its importance for geckos on islands was apparently masked by reduced gecko abundance in the presence of introduced predators, and suppression of host plants by introduced herbivores. Possible mechanisms of spread and new hosts of *C. zealandica* are described, and the vulnerability of the scale insect on islands with introduced mammals that suppress recruitment of selected host species is emphasised.

Keywords Scale insects, *Coelostomidia zealandica*; parasites; honeydew; geckos, *Hoplodactylus maculatus*, *Hoplodactylus duvaucelii*; hosts; ngaio, *Myoporum laetum*; karo, *Pittosporum crassifolium*; habitat modification; restoration.

INTRODUCTION

Island archipelagos show levels of recent extinction comparable to the most severe mass extinctions recorded in the Earth's history (Paulay 1994). These losses are largely the result of direct or indirect human activity, including the accidental or deliberate introduction of a large number of alien organisms. For example, in New Zealand over 1600 species of plants, 1500 species of insects and 90 species of vertebrates are established alien species (Atkinson and Cameron 1993), and some of them are now serious weeds and pests. Among the vertebrates, most extinctions of island plants and animals are attributable to predation and browsing by about nine species of mammals, including humans (Atkinson 1989). Two of the most widespread mammals are rabbits (*Oryctolagus cuniculus*) and Pacific rats or kiore (*Rattus exulans*). The effects of these two species on island systems have not been well documented. In an extreme example, rabbits appeared responsible for the loss of three species of endemic birds and 22 species of plants from Laysan Island (Williams *et al*. 1995 and references therein), but around New Zealand they more commonly induce low-diversity browse-resistant vegetation (Ogle 1990; Towns *et al.* 1997). Over the last 15 years, rabbits and Pacific rats have been removed from at least 24 islands on the continental shelf of New Zealand (Veitch 1995 and unpublished data). The islands cleared of these organisms provide opportunities to determine the direct and indirect effects of the introduced species and to restore depleted island systems. For example, recent studies on islands from which Pacific rats were removed indicated pervasive direct effects while the rats were present. These included suppressed recruitment of selected woody plants (Campbell and Atkinson 1999), reduced diversity and abundance of large, ground-dwelling flightless invertebrates such as weta (Orthoptera), some spiders and darkling beetles (Atkinson & Towns 2001), reduced capture frequencies, abundance and diversity of lizards (Towns 1994, 1996), impaired recruitment of tuatara (Tyrrell *et al.* 2000) and reduced productivity of small burrowing seabirds (Pierce 1998).

In addition to direct effects, invasive species are likely to have other effects that ripple through communities as changes in the populations of one species affect others (Simberloff 1990). For example, goats (*Capra hircus*) that reduce vegetation cover leading to increased light levels may also make sites prone to the effects of additional invasive weeds (Towns *et al.* 1997). Ripple effects may also lead to further pressure on indigenous species. Atkinson (1989) described modification of forest by introduced herbivores, leading to increased exposure of New Zealand land snails to introduced predators.

Ripple effects need not operate in one direction as a spiral of degradation. In theory, the removal of a catastrophic disturbance event could lead to ripple or interactive effects in the course of recovery or succession. Because the pre-disturbance history of many sites is unknown, the responses by resident species may sometimes be either unpredictable or unpredicted.

In this account I describe a hitherto unknown plant-scale insect relationship that was revealed following the removal

Pages 329-335 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

of rabbits and Pacific rats from islands in north-eastern New Zealand. The relationship is between a scale insect that exudes honeydew, the margarodid *Coelostomidia zealandica*, its host plants, and vertebrates that feed on the honeydew. This example illustrates how human-induced disturbance, and the presence of introduced mammals, can suppress populations of scale insects, their host plants, and the links to vertebrate honeydew feeders.

STUDY AREAS

The study was based on Korapuki Island and neighbouring Green and Middle Islands in the Mercury Islands, New Zealand (36°40'S, 175°52'E), a group of seven islands and associated islets and stacks of volcanic origin. Korapuki Island (18 ha) was inhabited by rabbits and Pacific rats until the rats were eliminated using rodenticide in 1986, and the rabbits by shooting in 1987 (Towns 1988; McFadden and Towns 1991). Vegetation on Korapuki Island was, until 1987, dominated by flax (*Phormium tenax*), shrubs including mahoe (*Melicytus ramiflorus*) and a canopy of pohutukawa (*Metrosideros excelsa*) consistent with extensive forest clearance (probably by burning early in the 20th century) and subsequent browsing of regrowth by rabbits. Since removal of rabbits, there have been spectacular increases in the recruitment of soft-leaved understorey species as well as increased canopy development of species such as mahoe (Towns *et al.* 1997).

Green (3 ha) and Middle Islands (13 ha) have never had introduced mammals. The islands support coastal broadleaf forest with little evidence of previous human occupation (Towns *et al.* 1990).

STUDY ORGANISMS

Myoporum laetum *(ngaio)*

Ngaio is a light-demanding shrub or tree that is often low-growing in coastal areas, but can reach to 10 m with a trunk diameter of 30 cm (Allan 1961). The species was regarded as uncommon on Korapuki Island by Atkinson in 1962 (cited in Hicks *et al*. 1975), but in 1974 a stand of ngaio was present (Hicks *et al*. 1975). When rabbits were eradicated in 1987, there were two identifiable stands of ngaio and occasional scattered trees on cliffs. The total number of trees was estimated as less than 10 by Towns *et al.* (1997) and the largest tree had a basal circumference of 1.25 m (unpublished data).

Pittosporum crassifolium *(karo)*

Karo is a shrub or tree that can reach 9 m and, although found in forest margins and stream sides (Allan 1961), also inhabits coastal areas, including small rocky islets. The species is able to germinate in very low light conditions. On Green and Middle Islands, karo is a significant component of the understorey as well as an emergent tree near the coast (Atkinson 1964; Cameron 1990; pers. obs.). Karo was ranked as frequent, with plants seen singly or in patches on parts of Korapuki Island in 1962 (Hicks *et al*. 1975), but was rare in the understorey by the time rabbits were removed in 1987 (Towns *et al.* 1997).

Coelostomidia zealandica

Ten native species of plant-sucking scale insects in the family Margarodidae are associated with native trees and shrubs. Margarodids produce sugary secretions (honeydew) that are used as a food source by forest insects and birds. The secretions also provide a medium for sooty moulds that blacken the trunks of heavily infested host plants (Morales 1991). The life history of *C. zealandica* is unknown, but that of a related species, *C. wairoensis*, was summarised by Morales (1991). In brief, the female life cycle includes a mobile crawler, non-mobile intermediate feeding stages, and a fully legged, mobile, non-feeding, flightless adult. The male life cycle includes a mobile crawler, a non-mobile intermediate feeding stage, non-feeding pre-pupa, pupa and fully-winged adult male.

Female *C. wairoensis* may remain and deposit eggs in a hard 'test' formed on its host, but females of *C. zealandica* are free living and oviposit in the soil or under bark (Morales 1991). Crawlers settle in cracks on branches, insert their mouthparts, and produce a long anal tube to void sugary waste. The other visible sign of infestation of hosts are white cocoons spun by male prepupae on the trunks of trees. Hosts include a wide variety of coastal and forest shrubs, trees, and vines. Amongst these are ngaio and *Pittosporum tenuifolium* (Morales 1991).

METHODS

Observations of the distribution and abundance of vegetation infected by margarodids were made on Korapuki Island opportunistically between 1986 and 2000 and on Lady Alice Islands and adjacent islets between 1992 and 2000. Infected trees were identified by the presence of sooty moulds, anal tubes of margarodid nymphs and, on some trees, by the presence of cocoons and adult males and females. Species identification was confirmed by R.C. Henderson (Landcare Research, New Zealand Arthropod Collection).

On Korapuki Island, counts of geckos on trees were made by capturing the animals after sunset (between 2200 and 2300 hrs) and marking them with a dot of correcting fluid (TWINK). Estimates of relative density on beaches were obtained from sightings within a fixed period of 'catch per-unit-effort' (CPUE) at five stations along a set transect line parallel with the high tide line (Towns 1991). The CPUE transect line was adjacent to the smaller of the two stands of ngaio. Comparative data were obtained from mammal-free Middle Island.

Levels of scale infestation were estimated by searching for deformations of the bark of karo and visible anal threads with honeydew droplets on karo and ngaio. Thread density was estimated by counting the active threads (those with droplets) within a defined 20 cm length of plant stem beginning 20 cm from ground level and continuing at 20 cm intervals (five per plant). The start of each counted cylinder was therefore 40 cm from the start of its predecessor. Thread density was calculated from stem circumference measured at the beginning of the counted sections and converted to threads per square metre.

RESULTS

Spread of ngaio and karo

Before removal of Pacific rats and rabbits from Korapuki Island, the remaining ngaio were infected by scale insects, and blackened by sooty mould. There was no visible recruitment of young ngaio to these populations. Likewise, karo was uncommon; there were no plants around the coastal beaches and apparently complete recruitment failure (Towns *et al*. 1997).

Following the removal of mammals, ngaio proliferated over most of Korapuki Island, but the seedlings and saplings failed to survive unless they occupied a canopy gap. The most rapid expansion was around the north-western coast of the island where ngaio formed almost continuous cover behind rocky beaches.

Likewise, after removal of mammals, karo proliferated throughout the island to become abundant on the coast and throughout forested areas (Towns *et al.* 1997). In coastal sites karo and ngaio formed mixed stands.

Spread of geckos

The two resident species of geckos, common gecko (*Hoplodactylus maculatus*) and Duvaucel's gecko (*H. duvaucelii*), were regarded as rare during surveys conducted on Korapuki Island in the presence of rabbits and Pacific rats (Whitaker 1973; Hicks *et al.* 1975; Towns unpublished data). CPUE data obtained in 1985, 1990, and 2000 indicated a 28-fold increase in sightings of common geckos along a coastal transect when comparing data from a year before campaigns against introduced mammals began (1985) against data from 13 years after the campaigns' completion (2000) (Fig. 1). Similarly, counts of geckos on flax flowers between 1986-1990 increased from 0% to 24% occupancy (Towns 1994). Both sets of data indicated an expanding population of geckos.

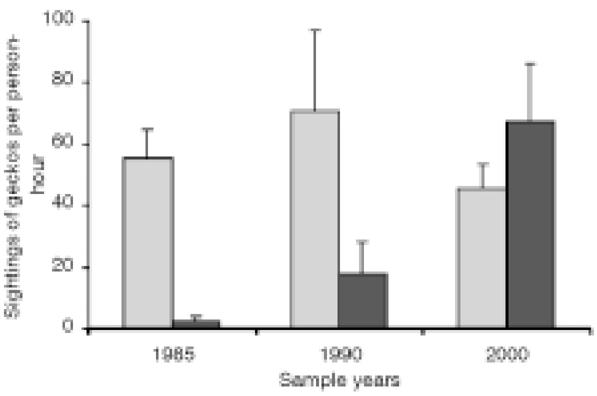

Fig. 1 Mean numbers (+ SE) of geckos sighted in November at five sites along a 100 m coastal transect on Middle Island (pale shading) and Korapuki Island (dark shading) one year before the start of campaigns against introduced mammals on Korapuki Island and in years subsequent to the completion of the campaigns.

Margarodids on Korapuki Island and nearby islands

By 1994, eight years after the beginning of the campaigns against rats and rabbits on Korapuki Island, young ngaio trees on the coast adjacent to the original ngaio stands were infested with margarodid scale insects. In addition, the margarodids had infested karo trees growing beneath the original ngaio.

On karo, infestations were in the form of scattered raised wart-like growths on the bark with the margarodid anal tubes projecting from these growths. Occasionally, lower branches were buried in leaf litter, and on these were external encrustations of scale. Of 20 karo checked within a ngaio stand in December 2000, 19 (95%) were infected by scale. These included saplings only 50 cm tall.

Unlike the localised and scattered infestations of scale on karo, those on ngaio were present over the entire trunk. In December 2000, mean infestations on five young trees (range 17-69 cm basal circumference) ranged from 70-2200 threads/m^2. The most heavily-infected tree had a thread density of up to 3300/m^2.

Margarodid infestation of karo and ngaio plants checked on Green and Middle Islands indicated similar levels of infestation to those found on Korapuki Island. On Green Island, ngaio and karo were infected and large karo trees (basal circumference >50 cm) had deformed sections of bark up to 5 cm across as evidence of previous margarodid presence. On some trees the deformed areas had cracked open and were oozing sap. On Middle Island, evidence of scale was present on 24 of 25 (96%) karo trees checked along 400 m on or within 50 m of the coast (mean basal karo circumference ± SE: 45.06 ± 5.48 cm).

Few adult margarodids were seen on Korapuki Island when trees were inspected in late spring (November-December), but males and females were seen frequently during late summer (February-March). Adult females were orange-pink in colour and were visible during daylight slowly crawling about on the bark of ngaio and on the leaf litter beneath ngaio trees. Females were seen once on karo trees.

Invertebrates and geckos on honeydew sources

Introduced *Vespula* wasps (probably *V. germanica*) were present on Korapuki Island until about 1990, but have not been seen since. Other introduced wasps include the Asian paper wasp *Polistes chinensis* and the Australian paper wasp *P. humilis*. Neither species has been seen accumulating around honeydew sources on ngaio. Two native insects were commonly active on ngaio at night: a cockroach (probably *Parellipsidon lattipennis*) and a byrrhid beetle (*Pedilophorus crysopepsis*). During the day, the native ant *Monomorium antarcticum* was seen on some trees and there were also occasional trails of the introduced ant *Technomyrmex albipes*.

Geckos were first seen aggregating on ngaio trees in March 1992. Both common and Duvaucel's geckos were observed at night on ngaio. Up to 11 Duvaucel's geckos and up to eight common geckos were observed simultaneously on individual ngaio trees. Both species were observed licking at the bark surface. An estimate of numbers of geckos using individual trees was obtained over six nights in December 1996. Of two trees checked, one had 10 common geckos and two Duvaucel's geckos over a total stem length of 3 m, and the other had 32 common geckos and one Duvaucel's gecko over a stem length of 7 m. Excluding recaptures, the number of common geckos observed on the two trees over six nights was 0.43 geckos/m/hour and 0.76 geckos/m/hour respectively.

DISCUSSION

The ecological role of honeydew

Congregations of geckos feeding on exudates from margarodids on ngaio trees have not previously been reported. There was little chance of observing the phenomenon before removal of introduced mammals from Korapuki Island because at that time geckos were rarely seen. Indeed, there is little published information on the use of honeydew by geckos anywhere in New Zealand, although the attraction of trees bearing honeydew to geckos has been exploited when surveying for geckos (A.H. Whitaker pers. comm.). Whitaker (1987) found at least 80 geckos (*Hoplodactylus pacificus*) feeding on honeydew on a 6 m tall karo on rodent-free Little Ohena Island. This is the only report now attributable to honeydew produced by *Coelostomidia zealandica*. Bishop (1992) included geckos in a diagrammatic representation of the users of honeydew in beech forest of the South Island, but gave no further details. Beggs (2001) included lizards among species that consume honeydew in beech forests of the South Island, but did not elaborate.

Honeydew can provide a major energy source in forest ecosystems (Beggs 2001). The most intensively studied honeydew-producing system in New Zealand is that of *Ultracoelostoma* spp. on black beech (*Nothofagus solandri* var. *solandri*), mountain beech (*N. solandri* var. *cliffortioides*) and red beech (*N. fusca*) in the northern South Island. The honeydew produced is rich in fructose, sucrose, glucose, and oligosaccharides, but low in protein (Grant and Beggs 1989) and is sufficient to provide the daily energy requirements of kaka (*Nestor meridionalis*), a large native parrot, after only three hours of feeding (Beggs and Wilson 1991). The sooty moulds that grow on the honeydew are also a food for arthropods (Morales *et al*. 1988), and there is evidence from northern hemisphere studies that honeydew washed into soil promotes the growth of microorganisms and these in turn affect processes such as carbon throughfall and nitrogen flux (Beggs 2001 and references therein).

Two species of Margarodidae were probably once widespread on the northern offshore islands: *Coelostomidia wairoensis*, most commonly on kanuka (*Kunzea* spp.), and *C. zealandica* on a variety of hosts including ngaio and

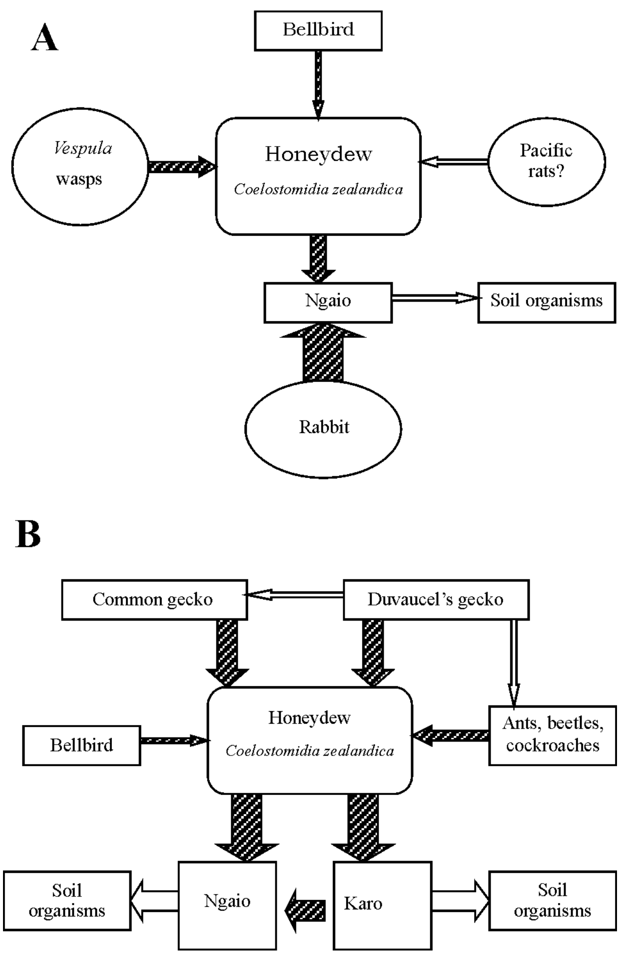

Fig. 2 Conceptual models of interactions between honeydew, their host trees and honeydew feeders before (A) and after (B) the removal or disappearance of introduced mammals and wasps (circles) from Korapuki Island. Arrow width denotes likely interaction effects; unshaded arrows are for potential rather than observed effects.

karo. Kanuka can form large stands in regenerating forest, such as those on Whatupuke and Lady Alice Islands in the Marotere Islands (Bellingham 1984). Duvaucel's geckos were observed on Whatupuke Island by A.H. Whitaker and R. Parrish (pers. comm.) feeding on honeydew exudates on kanuka. The combined output of honeydew from scale insects on kanuka, ngaio, and karo may therefore once have been considerable on offshore islands.

The most frequently observed vertebrates attracted to honeydew produced by *C. zealandica* were geckos (Fig. 2), although the honeydew also appeared attractive to nectivorous birds. For example, on Korapuki Island and in the Poor Knights Islands, bellbirds (*Anthornis melanura*) were seen foraging on ngaio and karo trees, presumably feeding on the exuding honeydew produced by *C. zealandica* (pers. obs.). Similarly, tui (*Prosthemadera novaeseelandiae*) were observed on Muriwhenua Island in the Marotere Islands foraging on ngaio infested with honeydew scale (Towns and Parrish unpublished data).

Unlike flowering plants, which often only provide seasonal nectar sources, honeydew has the advantage of being available continuously, although whether there is seasonal variation in the quantity and sugar concentration of honeydew produced by *Coelostomidia zealandica* is unknown. In beech forest, the composition and production of honeydew available varies during the year (Gaze and Clout 1983), but declines in honeydew production during late summer and autumn were probably due to consumption by introduced wasps (Moller and Tilley 1989). Activity by geckos is temperature dependent (Angilletta *et al.* 1999), so geckos may only feed on honeydew during the warmer months. There are no such constraints on birds such as bellbirds and tui.

Implications for restoration on islands

New Zealand margarodids are either quite host-specific or polyphagous. The most polyphagous species is *C. zealandica* (Morales 1991). However, only two of its previously reported host species commonly grow on the smaller northern offshore islands (less than 100 ha): the creeping vine *Muehlenbeckia* spp. and ngaio. The hitherto unreported capability of *C. zealandica* to infest karo provides a third host on these islands. Karo may also play a vital role in the scale insect's transmission. Unlike ngaio and *Muehlenbeckia*, karo is able to survive in low light conditions under the forest canopy. On Korapuki Island, despite the small number of parent trees, regeneration by karo was rapid and widespread (Towns *et al.* 1997). Since adult female margarodids have limited powers of dispersal, progressive infestations of karo have apparently enabled the spread of margarodids through some of the gaps between new stands of ngaio (Fig. 2). As a corollary, the absence of karo (and perhaps other hosts) from low-light areas could break the infestation pathway. Without removal of rabbits from Korapuki Island, the margarodids on ngaio would have eventually lost their original host trees, as they became overtopped by an expanding canopy of mahoe.

Regeneration of karo is strongly inhibited by Pacific rats (Campbell and Atkinson 1999), so it is conceivable that the presence of these rats alone may be sufficient to initiate the disappearance of *C. zealandica* from some islands. For example, no evidence of margarodids was found on regenerating karo eight years after the removal of Pacific rats from Red Mercury Island (I. A. E. Atkinson pers. comm.). Similarly, following the eradication of Pacific rats from Lady Alice Island in October 1994 (K. Hawkins pers. comm.), surveys for margarodids on extensive ngaio on the western coast failed to reveal visible signs of infestation by margarodids (unpublished data). There were no visible anal tubes and the bark of all trees was clear of sooty moulds. Only three adult karo trees are known to have survived on the island before removal of Pacific rats and these trees also appeared free of margarodids. By comparison, on islets such as Muriwhenua Island adjacent to Lady Alice Island, ngaio were heavily infested by margarodids. Muriwhenua Island also supports dense populations of common and Duvaucel's geckos (Whitaker 1978; Towns and Parrish unpublished data).

On the mainland, introduced organisms, especially wasps, can have devastating effects on the availability of honeydew for native species, reducing the standing crop of honeydew by 90% for five months of the year (Beggs 2001). On offshore islands, the threats to these resources appear to be more through the loss of infestation sources as a result of habitat destruction and modification of forest composition by introduced mammals (Fig. 2). Because the female margarodids are flightless, once the source of infestation is lost, there is no way for scale insects to reestablish, even when abundant host plants are present. This situation appears to prevail for *C. zealandica* on Red Mercury and Lady Alice Islands. Because of the implications that the presence of margarodids has for carrying capacity of invertebrates, geckos, and nectar-feeding birds, artificial spread of scale insects as part of island restoration projects now needs to be seriously considered.

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A strategy for Galapagos weeds

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Abstract Galapagos has a native vascular flora of some 500 species, 60 more species that are doubtfully native, and more than 600 introduced species. Introduced species are the most serious problem facing the native biota. The worst invasive plants are trees and other woody species, vines and grasses, and most of them were introduced deliberately. Many have invaded the Galapagos National Park and are also invasive in agricultural zones. A strategy for tackling the weeds problem includes prevention, control, eradication and restoration, the research required to develop and prioritise these management actions, and development of a legal framework for their implementation. Given limited resources for control, a risk assessment system for prioritising problem species and key sites is essential and is being developed. It will evaluate both species that are already present, and proposed introductions. A quarantine system for Galapagos has been designed and implementation commenced. Quarantine is essential if the balance between introduction and eradication is to be tipped towards the latter. Research includes investigations of the ecology and distribution of introduced plants, to determine factors (such as reproduction and dispersal rates and longevity of plants and their soil seedbanks) essential for the design of successful management programmes. Research on control techniques is also essential, since many Galapagos invasives are useful species that have not been subject to control elsewhere. Restoration research is beginning, focussing on methods of control combined with active restoration, such as seeding with native species. Invasive plants have only recently been widely recognised as high priority in Galapagos, and the first projects investigating the ecology of serious weeds are now yielding results. Control trials are leading to the development and adoption of effective field methods. Attempts have begun to eradicate species with still-small populations, but which are known as invasive elsewhere. A pilot project is also beginning, to assess the feasibility of eradicating a well-established invasive tree species. These measures, aside from their scientific and conservation value, also act as confidence builders, demonstrating to the public and land managers both the dangers of introduced species and the possibilities for their control and eradication.

Resumen Galápagos cuenta con una flora vascular nativa de algunas 500 especies, más 60 especies que son dudosamente nativas y más de 600 especies introducidas adicionales. Los organismos introducidos constituyen el problema más grave que enfrenta la biota nativa. Las peores plantas invasoras son árboles y otras especies leñosas, trepadoras y pastos, y la mayoría de las especies que están causando o podrían causar problemas fueron introducidas a propósito. Muchas de estas especies han invadido el Parque Nacional Galápagos, y son igualmente agresivas en las zonas agrícolas. Una estrategia para enfrentar este problema incluye prevención, control, erradicación y restauración, las investigaciones necesarias para desarrollar y priorizar estas acciones de manejo, y el desarrollo de un marco legal para su implementación. El primer paso es desarrollar medidas para priorizar los problemas, tomando en cuenta los recursos limitados para el control. Un sistema de evaluación de riesgos se está desarrollando, para evaluar tanto especies que ya se dan en las islas y para introducciones propuestas. Este último forma parte de la prevención: un sistema de cuarentena para Galápagos ha sido diseñado, y su implementación iniciado. La cuarentena puede reducir pero jamás parar las introducciones, pero es necesario para cambiar el equilibrio entre la introducción y la erradicación. El control y erradicación tienen dos componentes: investigación y manejo. Las investigaciones de la ecología y distribución de las plantas introducidas nos permiten determinar los factores necesarios para diseñar programas de control y erradicación que sean exitosos, tales como tasas de reproducción y dispersión, longevidad de plantas y semillas etc. La investigación para desarrollar nuevos métodos de control también se necesita, por lo que muchas especies invasivas en Galápagos son especies útiles y no han sido sujetos del control en otras partes. La investigación para la restauración ya empieza, con su enfoque en combinar acciones de restauración positivos en combinación con el control, tales como siembra de especies nativas. El programa de plantas introducidas en Galápagos esta creciendo rápidamente, por lo que las plantas invasoras han sido solo recién ampliamente reconocidas como de alta prioridad. Los primeros proyectos para investigar la ecología de las malezas más graves ya son produciendo resultados. Los ensayos de control llevan al desarrollo y uso de métodos de campo eficientes. Se han iniciado intentos para erradicar especies aun representadas únicamente por pequeñas poblaciones, pero las cuales están conocidas como invasoras graves en otros lugares. También ha comenzado un proyecto piloto para evaluar la factibilidad de erradicar un árbol invasora bien establecido. Estas medidas, aparte de su valor científico y para la conservación, además pueden aumentar la confianza y cambiar la opinión general sobre el peligro de las especies introducidas y la factibilidad de su control.

Keywords Environmental weeds; strategic planning; Galapagos; prevention; control; eradication; restoration; research; islands.

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THE PROBLEM

Galapagos is an isolated oceanic archipelago of volcanic islands lying 1000 km west of Ecuador, straddling the equator (Fig. 1). The date of discovery of the archipelago is usually recognised as 1535. The islands were uninhabited at that time, and no evidence proving earlier human presence has been found (Slevin 1959; Hickman 1985). The first visitors after discovery were mainly buccaneers, passing sailors, whalers and sealers (Hickman 1985). Settlement began on Floreana Island in the early 1800s, but Santa Cruz Island was only settled as late as the 1920s (Slevin 1959; Schofield 1989).

The pirates and whalers deliberately or accidentally introduced some alien species, including goats, rats and, probably, insects and plants. Even before permanent settlement, Floreana had large areas dominated by introduced plants such as *Citrus* spp. (Slevin 1959; Hamann 1984). The rapidly increasing settled population, growing at 8% per year in the 1990s through both immigration and births, has been accompanied by an enormous number of new introductions of alien plants and animals (Mauchamp 1997). Although agricultural development began at the time of settlement, the process has been uneven, leading to different rates of introduction of alien species. Floreana has the longest history of the presence of a large introduced flora, while agriculture on Santa Cruz was minimal until about 1960 (Moll 1990).

The Galapagos National Park forms 96.4% of the land area of Galapagos; inhabited areas (urban and agricultural zones, military bases and airports) make up the rest. Alien plants that have escaped from cultivation are mostly found on the five inhabited islands, especially the four with agricultural and urban zones (Floreana, Isabela, San Cristóbal, Santa Cruz: Fig. 1); the fifth, Baltra Islet off the north coast of Santa Cruz, is a military base and civil airport. There are also a number of deliberately-introduced species on Santiago, which was formerly inhabited. Aliens that were introduced accidentally often have a much wider distribution in the archipelago but are mostly less problematic than deliberately introduced species. Most introduced plant species are found in the more humid, higher altitudes of the four larger inhabited islands, and the settled areas are the major source of invasion into the Galapagos National Park.

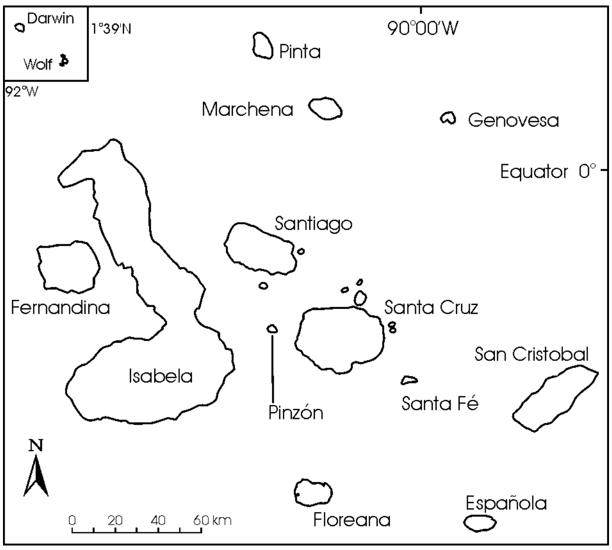

Fig. 1 Galapagos archipelago.

The Galapagos islands support a native vascular flora of about 500 species, with an additional 60 doubtfully native species, principally pantropical ruderals, which may have arrived naturally or may have been introduced by the earliest human visitors to the islands. In this paper, "introduced" and "alien" are used interchangeably to mean introduced deliberately or accidentally due to the actions of man. "Naturalised" means reproducing in natural or semi-natural habitat without the further assistance of man (beyond habitat disturbance). "Invasive" means invading natural (undisturbed) habitats. "Weed" means a naturalised species.

Porter (1822) mentioned the first alien species (pumpkins *Cucurbita* sp. and potatoes *Solanum tuberosum*), which were introduced about 1807. Numbers increased slowly until the 1960s, although true numbers are unknown, since earlier references (especially Wiggins and Porter 1971; Porter 1984) took into account only naturalised species. The list continued to increase, reaching 438 in 1995 (Mauchamp 1997) and over 600 by November 2000 (Database of the Galapagos Flora, Charles Darwin Research Station). The minimum detection rate has thus been more than 10 per year in the last 30 years (Fig. 2), and more than 120 during 2000. However, the recent apparent rate of increase is obviously affected by increased interest in recent years in the introduction process, as well as increased sampling effort and individual research projects, and the inclusion of cultivated, non-naturalised species in more recent lists (Tye 2001b).

The principal threat to the terrestrial biota of Galapagos is introduced species (Loope *et al.* 1988). Most (*c.* 75%) of the alien plant species were introduced deliberately as useful plants, for their ornamental, agricultural, medicinal or timber value, although some were introduced accidentally. An even higher percentage of the worst invaders was introduced deliberately (see Mauchamp 1997). Some 45% of introduced plant species have naturalised (A. Tye unpublished data).

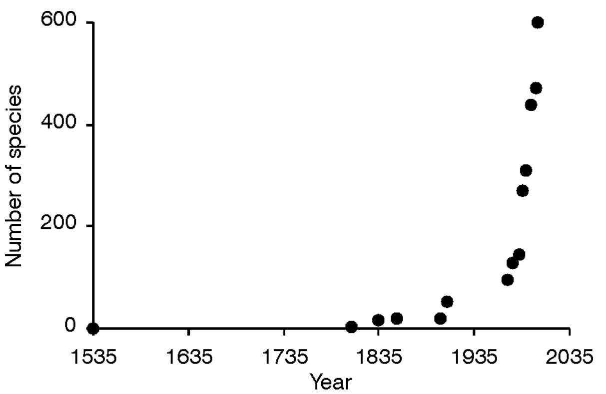

Fig. 2 Rate of increase in numbers of introduced plant species recorded in Galapagos since their discovery in 1535 (Tye 2001b).

Most introduced plant species have not significantly affected the ecological equilibrium of the islands. However, at least 37 species (Table 1) have invaded large areas and/or appear to be adversely affecting the natural ecosystem to a degree more than simply occupying space within an existing community (e.g. altering community composition or threatening a native species), or (in a few cases) are naturalised but with limited distributions and known to be extremely serious invasives in other parts of the world. At least 80 more species have naturalised and are still uncommon but are known or suspected to have invasive tendencies. Another 140 or so have naturalised but do not appear to be causing obvious ecological damage, while at least another 100 species present in the islands, which have not yet escaped from cultivation, are known to be seriously invasive in other parts of the world. The rest of the introduced species are present in cultivation; some may become invasive. Lawesson (1990) lists only eight serious invasives, while Mauchamp (1997) lists 11, omitting one of those included by Lawesson. The 37 species in Table 1 include these 12, the increase largely caused by a re-evaluation of the threat posed by certain species (four species listed by Mauchamp 1997 as potential invaders are included in Table 1), as well as a few recently-added species.

Table 1. Invasive species in Galapagos that are known or suspected to be causing significant ecological change, including in natural areas.

Family	Species
Agavaceae	*Furcraea hexapetala* (Jacq.) Urb.
Aristolochiaceae	*Aristolochia odoratissima* L.
Asteraceae	*Pseudelephantopus spicatus* (B. Juss. ex Aubl.) C.F. Baker
Bombacaceae	*Ochroma pyramidale* (Cav. ex Lam.) Urb.
Boraginaceae	*Cordia alliodora* (Ruiz & Pav.) Oken
Caesalpiniaceae	*Caesalpinia bonduc* (L.) Roxb.
	Senna obtusifolia (L.) H.S. Irwin & Barneby
Capparidaceae	*Cleome viscosa* L.
Crassulaceae	*Bryophyllum pinnatum* (Lam.) Oken
Cucurbitaceae	*Cucumis dipsaceus* Ehrenb. ex Spach
Euphorbiaceae	*Dalechampia scandens* L.
	Ricinus communis L.
Lamiaceae	*Hyptis* cf. *atrorubens* Poit.
Lauraceae	*Persea americana* Mill.
Meliaceae	*Cedrela odorata* L.
Mimosaceae	*Leucaena leucocephala* (Lam.) de Wit
Myrtaceae	*Psidium guajava* L.
	Syzygium jambos (L.) Alston
	Syzygium malaccense (L.) Merr. & L.M. Perry
Passifloraceae	*Passiflora edulis* Sims.
Poaceae	*Digitaria decumbens* Stent
	Melinis minutiflora P. Beauv.
	Panicum maximum Jacq.
	Pennisetum clandestinum Hochst. ex Chiov.
	Pennisetum purpureum Schumach.
	Urochloa brizantha (Hochst. ex A.Rich) R.D.Webster
	Urochloa mutica (Forssk.) T.Q. Nguyen
Rosaceae	*Rubus niveus* Thunb.
Rubiaceae	*Cinchona pubescens* Vahl
Rutaceae	*Citrus aurantiifolia* (Christm.) Swingle
	Citrus limetta Risso
	Citrus limon (L.) Bur.f.
Solanaceae	*Cestrum auriculatum* L'Hér.
	Datura stramonium L.
	Solanum lycopersicum L.
Ulmaceae	*Trema micrantha* (L.) Blume
Verbenaceae	*Lantana camara* L.

There have been few rigorous studies of the effects of the invasions, but some species have caused drastic habitat changes, forming monospecific stands, shading out or otherwise replacing native vegetation communities, or preventing seedling regeneration by forming impenetrable carpets. Where detailed studies have been made, dramatic community changes have been revealed (Jäger 1999). The worst effects seem to be caused by woody species, especially trees such as *Psidium guajava*, *Cedrela odorata* and *Cinchona pubescens*, and bushes that form impenetrable thickets, such as *Lantana camara* and *Rubus* spp. Many vines and grasses are also causing serious problems.

Although most of the serious invaders in Galapagos were introduced deliberately as useful plants, most of the species that are problematic in the National Park are also causing problems for farmers, and this includes cultivated species as well as accidental introductions. To a large extent, therefore, there is little conflict regarding priorities for control or eradication of invasives. However, this is not true in every case. For species such as *Cinchona pubescens* (which produces no economic yield of quinine or wood), all parties wish to see the pest eradicated, whereas *Pennisetum purpureum* is a serious invader of the National Park but is a valued pasture grass and its removal from agricultural areas would be politically and economically difficult. In such cases, an eradication strategy would have to include replacement by a non-invasive substitute.

THE STRATEGY

The strategy for dealing with invasive plants comprises five levels of action: (1) prevention, (2) control and eradication, (3) restoration, (4) the research required to prioritise and develop appropriate techniques to carry out these actions, and (5) development of a legal framework for their effective implementation.

Research

Prioritisation

Given limited resources for control, an essential first step is prioritising the problems. A risk assessment system is used to evaluate both species that are already present in the islands and proposed introductions. Permitted lists for import of species and products have already been drafted, based on a preliminary risk assessment procedure, while all introduced plant species have been subject to a subjective risk assessment. A formal risk assessment system is currently being developed (Tye 2001a). Risk assessment and prioritisation will also be applied to sites. This is already done in a subjective manner, but a more formal system based on conservation value is being developed.

Risk assessment is the major tool used to decide whether an introduced species should be merely controlled, or whether an eradication attempt could be considered. The decision depends on the plant's inherent biological characteristics (including growth form, reproductive strategies, dispersal mechanisms, reproductive rate, age at first reproduction, and longevity), its current distribution, abundance, rate of spread, ecological and other impacts, susceptibility to treatment (visibility, availability of control techniques cheap and effective enough to enable eradication to be contemplated with available resources), and urgency of action required.

Ecology and distribution

The success of a risk assessment system depends on sound scientific data about the distribution, biology, and ecology of the introduced species. These factors are also essential to the design of control or eradication programmes. Research currently in progress or planned for the next five years includes complete surveys of introduced plants in the agricultural and urban zones of the four major inhabited islands, ecological studies of the life cycle of the worst invaders and their effects on native vegetation, and gathering of information on the biology of introduced species from sources outside Galapagos. So far, ecological and distributional studies have focussed on *Cinchona pubescens* (Jäger 1999), *Psidium guajava* and *Rubus* spp.

Control trials

Research on control techniques is also essential, since many Galapagos invasives are useful species and have not been subject to control elsewhere. We need effective methods of killing the invaders that are both as cheap as possible and are appropriate for use within a national park, causing minimal damage to the native vegetation. Limitations are imposed by the unavailability of some herbicides in Ecuador and by the difficulties of obtaining, using, and servicing sophisticated application equipment. The trials take these limitations into account, although where a herbicide is not locally available (but seems by far the most effective), trials are undertaken along with negotiations with potential suppliers and the Ecuadorian authorities in order to try to ensure that the required materials become available for use within the archipelago. Research includes trials of manual methods, herbicides, and herbicide application techniques. Trials so far have included 15 of the worst invasive plant species. Biological control has never been applied in Galapagos and is controversial because of the potential dangers of introducing yet another organism, but laboratory research is currently under way in preparation for what may become the first biocontrol field trial, for cottony cushion scale insect *Icerya purchasii*. Should this first instance prove successful, the climate of opinion should improve, and trials for the biocontrol of weeds, including *Cinchona pubescens, Lantana camara,* and *Rubus* spp.,may be considered.

Given the pressure for control of the worst invaders, trial results are often applied to control and eradication campaigns before completion of trial monitoring. In such cases the control action is also monitored to enable better assessment of the efficacy of the techniques. Trial results are also fed into the weed risk assessment system, since availability of suitable techniques is an important factor in risk assessment, particularly in helping to decide whether a plant species should be controlled or whether an eradication attempt could be considered.

Restoration research

Restoration research is just beginning in Galapagos. The initial focus is on methods of control combined with active restoration, such as seeding with native species. Only one project to investigate the possibility of assisted restoration has so far been undertaken, examining restoration of native *Scalesia* forest following control of elephant grass *Pennisetum purpureum*. In other cases, monitoring of selected control sites is being undertaken, to identify when positive intervention to restore native vegetation communities following control is needed, and to design such projects. Two examples include monitoring of *Miconia* Zone vegetation following control of *Rubus niveus* and other introduced species at the unique highland crater-lake site of El Junco on San Cristóbal Island, and monitoring of regeneration following control of *Cedrela odorata* in Transition Zone forest on Santa Cruz Island. Given the limited resources available for introduced plant control in Galapagos in the foreseeable future, it is unlikely that major restoration projects will be able to be considered for some time.

Management

Prevention

Prevention is the first stage of management action: a quarantine system for Galapagos has been designed and its implementation commenced (Zapata *et al.* 2000). Quarantine can slow but never stop introductions, although it is essential if the balance between introduction and eradication is to be tipped towards the latter. The quarantine system will eventually implement primary control outside of Galapagos, at the mainland port and airports from which traffic reaches Galapagos. Secondary quarantine control is already in place on the islands, and an early warning and rapid response strategy is being planned for species that evade these controls. The system deals also with transport between the islands, given the importance of inter-island differences in biodiversity, with many single-island endemic species.

Control and eradication

Eradication is considered for plants at both ends of the invasion scale in Galapagos. One priority group comprises plants that are known to be invasive in other parts of the world but are present in Galapagos in very small populations and are not regarded as indispensable by the local community. At the other end of the scale are species that are seriously invasive already in Galapagos, but for which an assessment indicates that eradication might be feasible. We have two successful examples of eradication of plants in the first group and projects in progress to eradicate more, but are only beginning to consider eradicating plants in the second group.

Among the first group, plants known to be invasive but still present in small populations in Galapagos, two species are considered eradicated from Galapagos. *Echinopsis pachanoi* (Cactaceae) was known from a single garden plant on Santa Cruz Island, which was cut down and burned in the 1980s. The other species was tropical kudzu *Pueraria phaseoloides* (Fabaceae), which is widely grown in mainland Ecuador for ground cover in orchards and as a forage plant and soil improver. It was introduced by one farmer in 1996 and planted in a single pasture field. The plant was spotted by a CDRS botanist, and after the potential danger of the species was explained to the farmer (it is a close relative of kudzu *Pueraria lobata*, one of the worst invaders of the south-east U.S.A), he agreed to allow it to be eliminated. The plot was treated with glyphosate and monitored regularly, with new growth spot treated. No plants have been seen since 1997 although monitoring continued until 2001 (further details in Soria *et al.* 2002). Although considered eradicated, we still lack introduced species inventory data on the agricultural and/or urban zones of three of the four inhabited islands, so there may yet be other plants of these species present there.

In the past, apart from these two isolated cases, eradication has not been attempted. Instead, funds have been directed as a priority towards control of the most serious invaders in selected important sites. However, during the past year we have begun to attempt eradication on a larger scale. A project began in 2001, with sufficient funding to permit the selection and attempted eradication of 30 plant species still present in small populations. Eradication in some cases will be from the entire archipelago and in other cases only from selected islands. Eradication attempts have already begun with the following species:

- *Citharexylum gentryi*, a timber tree present in two parts of the agricultural zone of Santa Cruz.
- *Rubus megalococcus* and *R. glaucus*, each present in a single population on Santa Cruz.
- *Rubus niveus* from the island of Isabela. This species is already widespread and beyond current eradication capability on Santa Cruz and San Cristobal islands, but is present in only a small area on Isabela.
- *Rubus adenotrichos*, present in single localities on Isabela and Santa Cruz.

These cases are discussed in more detail by Soria *et al*. (2002). The remaining priority plants to make up the initial target 30 species will include other timber trees and ornamental plants. A preliminary list of some 60 potential targets has been drawn up based on the basic criteria of limited distribution and known invasive tendencies, and the final list of 30 will be selected following a risk assessment based on a full range of criteria including feasibility of eradication.

The second priority group of species for eradication comprises serious invaders that are widespread but whose characteristics suggest that eradication might still be feasible. The pilot species for this work is *Cinchona pubescens*, a small tree that has become the greatest single threat to the native highland vegetation of Santa Cruz. The principal characteristics suggesting that it might be feasible to control it include that it is conspicuous, not regarded by the local community as a useful species but on the contrary widely recognised among conservation workers and local people as a serious pest, and it is present on only one island. In addition, whereas previously it had been difficult or impossible to control *Cinchona* by acceptable herbicides, safe and effective chemical techniques for killing it, using picloram for cut stump treatment and picloram-methyl metsulphuron mixtures for hack-and-squirt treatment, have recently been identified by research in Galapagos. These herbicide treatments for larger trees, combined with manual removal of seedlings and saplings, could eventually permit complete eradication. This project is still in the planning stage, with larger-scale control trials under way and an ecological study in progress that is designed to provide information on factors affecting the speed and likelihood of eradication, such as density, distribution, seedbank longevity and dispersal ability. These studies will lead to better cost estimates, which are needed for the ultimate decision on whether to attempt eradication.

Control is undertaken where eradication is not currently considered feasible but where the plant is considered to pose a significant conservation risk. Such plants include most of the most serious invaders, such as *Rubus niveus*, *Lantana camara*, vines such as *Passiflora edulis*, and very widespread invasive trees such as *Psidium guajava* and *Cedrela odorata*. Virtually every widespread introduced plant that is not a tree also falls into this category, as do useful (indispensable to local people) but invasive species for which no effective substitute species has yet been identified. Some potentially invasive garden ornamentals are also considered not yet eradicable, since they are so popular.

Restoration

So far, no active restoration has been undertaken following invasive plant control. Monitoring of selected control sites will provide data that will be used to identify needs for intervention to restore native vegetation communities and to design such projects.

Legal framework

A comprehensive "Special Law for Galapagos Province" was passed by the Ecuadorian Government in 1998. It covers all aspects of conservation and development in the province (which includes the entire archipelago), from immigration and waste disposal to natural resource use, conservation of rare species, and control of pests. Regulations for the implementation of the new law are currently being drawn up, including sections on quarantine and introduced species management. Weed risk assessment will be written into these regulations, as will requirements for planning for the control or eradication of introduced plants. The regulations will place legal obligations on various public and private bodies, as well as on individual landowners, to undertake specified actions in regard to declared weed species or new introduced plants, including monitoring, reporting, controlling, and eradicating. Such regulations and obligations are especially essential for the success of eradication efforts, where all populations of the species, whether on public or private land, must be treated.

THE FUTURE

Recent advances in weed control in Galapagos have included the development of an objective risk assessment system, the first properly designed, implemented and monitored control trials, the first comprehensive projects to investigate the ecology of serious weeds, the first strategic eradication programme, the first attempt to eradicate a well-established invader, and the approval of the first comprehensive legislation to deal with the problem of environmental weeds. All of these developments have taken place in the last five years, and serious attempts to eradicate invasive species from the archipelago have begun to look promising. These activities, aside from their scientific and conservation value, should act as valuable confidence-builders, changing the climate of opinion about the dangers of introduced species and the feasibility of their control and eradication. If successful, we could be about to experience the first recorded decrease in the number of introduced plant species in Galapagos since their discovery in 1535.

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Eradicating Indian musk shrews (*Suncus murinus*, Soricidae) from Mauritian offshore islands

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Abstract The Indian musk shrew (*Suncus murinus*), an efficient and rapid coloniser, has spread from its original home in India to become an ecological threat of global importance. A project to eradicate musk shrews from a 25 ha Mauritian offshore island began in July 1999. Due to the shrew's low susceptibility to anticoagulant poisons, we relied on live trapping. Seven months of continual trapping initially appeared to have been successful but the population soon returned to its original level. A second experimental eradication on a smaller island (2 ha), carried out over three weeks in June 2000, allowed us to monitor the eradication process more closely, and a return visit has revealed no further signs of shrews. Studies of bait preference and trap use, in field and captive situations, gave further insights into how to attract shrews into traps. The invasive land snail *Achatina fulica* proved by far the most successful bait. Captive trap trials revealed design problems in the type of traps used in both eradication attempts, which resulted in one third of animals escaping capture.

Keywords Invasive species; live trapping; poisoning; bait selection; island eradications; impact on native species.

INTRODUCTION

Natural history of the Indian musk shrew

The Indian musk shrew (*Suncus murinus*) is a highly adaptable insectivore, one of the largest members of the family Soricidae. Morphologically it is extremely variable, with body weights ranging from 33.2 g to 147.3 g in males and from 23.5 g to 80.0 g in females (Ruedi *et al.* 1996). Believed to have originated in the Indian subcontinent (Yosida 1982), its native range stretches across southern Asia from Afghanistan to the Malay archipelago and southern Japan. It has since been introduced into northern and eastern Africa, as well as much of the Middle East (Ruedi *et al*. 1996). Although this species can be found in forests and agricultural land, it is particularly common around areas of human activity. This association has contributed to its passive transportation to a number of oceanic islands, including Guam in the Pacific (Peterson 1956), and Madagascar, the Maldives, and Mauritius in the Indian Ocean (Wilson and Reeder 1993). The first reliable records of musk shrews in the western Indian Ocean date from the early 19th century (Hutterer and Trainer 1990), but the species is believed to have been on Mauritius since around 1760 (Cheke 1987).

The Indian musk shrew as an invasive species

This species is now widespread and rapidly expanding its range, and represents a major ecological threat. Its commensal habit, combined with the prodigious capacity for reproduction common to many small mammal species, makes it a highly effective coloniser. Although nominally an insectivore, the musk shrew is an opportunistic feeder and in some areas is known to feed predominantly on plant material (Advani and Rana 1981). On Mauritius, the shrew is known to prey upon native and introduced invertebrates, as well as damaging seeds and young plants by digging. Through predation or competition, musk shrews are believed to have caused the extirpation of several species of endemic lizards from the mainland of Mauritius, Reunion, and many of their offshore islands (Jones 1988, 1993). Since their introduction to the nearby island of Rodrigues in 1997 they have colonised the whole island (approximately 109 km^2) and have been strongly implicated in a sharp decline in the numbers of several invertebrate species, including two native centipedes and a field cricket. Beyond the Indian Ocean, the shrews are also causing widespread ecological damage. Since musk shrews were introduced to Guam in the early 1950s, the extirpation pattern of one lizard species, the pelagic gecko (*Nactus pelagicus*), has coincided almost exactly with the spread of the shrews. To a lesser extent, they are also believed to have affected two skink species *Emoia cyanura* and *Emoia caeruleocauda* (Rodda and Fritts 1992; Fritts and Rodda 1998). The musk shrew is fast becoming a pest species of global proportions, especially in disturbed, fragile, or small island ecosystems. The development of an effective method of eradicating or controlling this species is now a conservation priority.

Unlike some commensal species introduced over a wide geographical range, such as rats (especially *Rattus rattus* and *R. norvegicus*) and house mice (*Mus musculus*), musk shrews are currently extremely difficult to control using poison. The second-generation anticoagulants such as brodifacoum, which are effective for controlling rodents, are relatively ineffective on shrews. Differences in feeding habits and susceptibility make it difficult to get shrews

Pages 342-349 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

to consume either acute-acting or anticoagulant poison in quantities large enough to kill them (Morris and Morris 1991; Bell and Bell 1996). Toxicity studies suggest insectivores are generally less susceptible to commercially available poisons than rodents and herbivores. The few results available suggest musk shrews are killed by doses of around 47 mg/kg of brodifacoum, about 10 times the lethal dose for moles and 100 times the level needed to kill rats (Morris and Morris 1991). In addition, since shrews are not agricultural pests on the scale of rats and mice, there is far less commercial pressure on agrochemical companies to develop compounds specific to them. However, unlike rodent pests, notably *R. norvegicus*, shrews are not known to exhibit neophobia and indeed readily explore novel objects (Churchfield 1990; Gurnell and Flowerdew 1990). A previous study of the shrews on Ile aux Aigrettes, an offshore Mauritian island, showed them to enter live traps in large numbers (Pilgrim 1996). Therefore, in the absence of an effective chemical method, we decided that trapping was the most appropriate technique to eradicate musk shrews from the small offshore islands.

The Mauritian offshore island shrew eradication projects

The Ile aux Aigrettes habitat restoration project

Ile aux Aigrettes (Fig. 1), a 25 ha designated nature reserve, is the site of a major habitat reconstruction programme under the control of the Mauritian Wildlife Foundation (MWF) and the Mauritian National Parks and Conservation Service (NPCS). The island is the subject of a management plan (Dulloo *et al.* 1997), describing the physical and biological character of the island, as well as a detailed description of past and future management objectives. Ile aux Aigrettes is a flat coralline island reaching a maximum height of 13 m. The surface of the island is covered with holes and pinnacles of jagged eroded coral, covered in places with shallow soil seldom more than 15cm deep. Despite this, it has the highest indigenous vegetation cover of all the inshore Mauritian islands. It contains the largest area of coastal lowland forest remaining in Mauritius, including species such as the critically endangered ebony (*Diospyros egrettarum*) as well as other endangered and vulnerable plant species. An intensive weeding programme is systematically removing invasive plant species, while a nursery situated on the island is producing native plants for replanting in the weeded areas. Following the eradication of feral cats (*Felis catus*) and rats (*Rattus rattus*), endemic pink pigeons (*Columba mayeri*) and Mauritian kestrels (*Falco punctatus*) have been reintroduced. It is also planned to introduce ecological analogues of extinct species. The first of these releases has just taken place, with an experimental introduction of Aldabran giant tortoises (*Testudo gigantea*) in place of the two extinct Mauritian species (*Geochelone inepta* and *G. indicus*). The next phase of the restoration plan is to establish several species of endangered endemic lizards, some of which are restricted to single island populations (all information in Dulloo *et al.* 1997). However, since shrews are known to both prey upon small lizards and compete with them for food (Jones 1988, 1993), it is necessary first to remove the musk shrews from the island. A second aim was to train staff from both MWF and NPCS in shrew trapping techniques. As a signatory to the Convention on Biological Diversity, Mauritius is pledged to the eradication or control of invasive species, but there is a severe shortage of people with expertise in invasive species control, especially in developing countries.

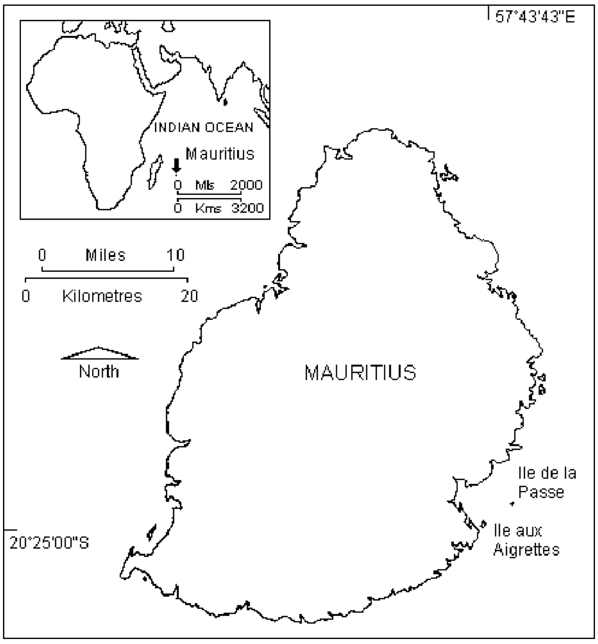

Fig. 1 Map of Mauritius showing position of Ile aux Aigrettes and Ile de la Passe (adapted from Bullock 1986).

Ile de la Passe

Following the Ile aux Aigrettes project we were keen to try our methods on a smaller and more manageable island, where the whole process could be monitored more closely and where we could manipulate trapping densities and trap coverage. An opportunity was presented when shrews were discovered on the nearby islet of Ile de la Passe (Fig. 1), some 4km north-east of Ile aux Aigrettes. This tiny windswept coralline island has an area of about 2 ha, less than one tenth the size of Ile aux Aigrettes, and is much less densely vegetated and topographically simpler. This removed the problems of size and accessibility we faced on Ile aux Aigrettes. In fact, the vegetation of Ile de la Passe was simple in the extreme, consisting almost entirely of short grass (*Stenotaphrum* sp.) and the occasional small bush of *Tournefortia argentea*. The outer edge of the island consists of exposed highly eroded jagged coral, but most of the island is smooth grassland, a habitat type not found on Ile aux Aigrettes. Few vertebrate species were present on the island. The shrews were the only resident mammals, while reptiles were represented by native Bouton's skinks (*Cryptoblepharus boutoni*), and introduced night geckoes (probably *Hemidactylus* sp.). The resident bird fauna consisted of a single pair of introduced house sparrows (*Passer domesticus*), although passing seabirds occasionally landed there. *Achatina* snails appeared to be absent, with no live animals encountered during three weeks on the island, and only one empty shell. This contrasts with the situation on Ile aux Aigrettes where live snails are common and empty snail shells are abundant across the island. This part of the project had two main aims. First, to learn more about shrew ecology, trapping, and the acceptability of different baits before restarting the Ile aux Aigrettes project. The second, and more immediate aim, was to protect a population of Bojer's skinks (*Gongylomorphus bojeri*) on a neighbouring island. This species is now restricted to six small Mauritian offshore islands, one of which, Ilot Vacoas, is only a few hundred metres from Ile de la Passe. MWF staff spent several days on this tiny island (less than 1 ha) in June 2000 and no signs of shrews or other introduced mammals were found. Shrews have been strongly implicated in the decline of Bojer's skinks on other islands (Jones 1993) and, since Ile de la Passe and Ilot Vacoas are potentially joined by a land bridge at very low tides, it was imperative to remove them from Ile de la Passe as soon as possible.

METHODS AND RESULTS

Indian musk shrew eradication projects

The Ile aux Aigrettes shrew trapping programme

A large-scale trapping programme began in July 1999. A 12.5 x 12.5 m grid was marked out across the island using blue polypropylene twine and traps were set at the intersections. This trap spacing was based on the findings of a mark-recapture study carried out on Ile aux Aigrettes by Pilgrim (1996), which showed that shrews travelled up to 60m between captures. Her study found that 15 x 15 m grids caught substantial numbers of shrews; we decided on a 12.5 x 12.5 m grid to increase trapping intensity. This gave a total of 1651 trap points. Longworth traps and plastic tunnel traps of a similar design ('Trip traps' (Fig. 2), manufactured by Proctor Bros. Ltd., Pantglass Industrial Estate, Bedwas, Caerphilly, Wales, CF83 8XD) were used

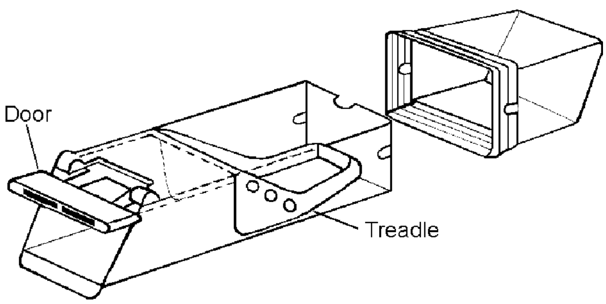

Fig. 2 A 'set' Trip trap. When set, the door rests on the treadle in a horizontal position. The bait is placed in the end section of the trap, and the two sections clip together. Animals entering the trap press down on the treadle, releasing the door, which falls down into a vertical position behind them. When assembled the trap measures 40 x 50 x 165 mm.

for the first three months (*c*. 50,000 trap nights). Trip traps alone were used for the remainder of the programme (*c*.50,000 trap nights). Since trap numbers were never sufficient to cover the whole surface of the island, traps were moved across the island as a rolling front. Starting at the western end of the island, up to two thirds of the island was initially covered with traps. A maximum of 1100 traps were in use at any one time. For the first three sweeps the traps at the furthest west part of the island were lifted and moved to the eastern side of the block of traps after 4-15 days. Thereafter trapping periods were more variable depending on capture rate. This process was repeated for six 'sweeps' of the island. Traps were baited with a variety of substances to try to maximise attractiveness to the shrews. Sweep one used rehydrated dried fish mixed with flour and vegetable oil; sweep two rehydrated fish mixed with vegetable oil; sweep three soaked sultanas; sweep four rehydrated dried fish mixed with cod liver oil; and sweeps five and six a mixture of peanut butter and oats.

Initially the programme appeared very successful and the number of captures declined asymptotically in a classic extinction curve. For over six weeks no captures were made, but at the end of November 1999 shrews began to be encountered once more. Captures continued at a low but steady rate for the next three months, and shrews were caught wherever traps were placed. Table 1 shows the capture rate per 100 trap nights of each of the six trapping sweeps. The shrews caught in sweeps five and six included the first pregnant and lactating females caught during the programme. Of the 54 shrews caught during the last three months, 76% of the females were pregnant and/or lactating. The trapping programme was discontinued at the end of February 2000, when it became apparent that shrews were present wherever traps were placed and insufficient traps were available to cover the whole island. Weather conditions were also affecting the efficiency of the traps, with the percentage of traps found tripped but with no capture increasing from around 4% in August to 23% in January and February during the rainy season. Within months, shrews were as abundant as they had been before the trapping programme began, testament to the species' phenomenal powers of reproduction. Summary morphometric statistics for this group of shrews, and those caught in subsequent sections of the study, are shown in Appendix 1.

The Ile de la Passe shrew trapping programme

This trapping programme ran over 20 days in June 2000. We divided the island into four approximately equal sections and used traps baited with a different substance in each section. These baits (cheese, dog food, sardines and mayonnaise) were moved every five days, so after 20 days

Table 1 Capture rate (shrew captures per 100 trap nights) for sweeps one to six.

Sweep	1	2	3	4	5	6
Capture rate	5.89	0.15	0.02	0.02	0.11	0.15

each bait had been offered in each section for five days. This strategy was designed to pick up any stragglers who might have shown aversion to one bait type. In the event, all but one of the shrews were caught in the first five days, so the bait preference component was never tested. Forty shrews were caught in total. During a return visit made to the island in October 2000, 600 further trap nights revealed no sign of shrews. Night walks and careful searches for droppings, focusing on the buildings in which shrews had initially been concentrated, also proved negative. Small-scale ecological changes were apparent on this second trip, with one species of large cockroach being particularly abundant. It is likely that these invertebrates formed a substantial part of the shrews' diet and that their subsequent increase is due to this reduction in predation pressure. Further checks are planned after another four months and then at yearly intervals. However, so far we are cautiously optimistic that we have carried out the world's first successful eradication of musk shrews from an island.

Comparison of the Ile aux Aigrettes and Ile de la Passe projects

The two projects differed in a number of respects, summarised in Table 2 below. The most notable differences are the length of the project and the size of the islands, both about an order of magnitude greater on Ile aux Aigrettes. Other differences with the Ile de la Passe project were that the trapping density was about twice as high and that the traps were set simultaneously over the entire island for the duration of the trapping programme. The shrew density was about 50% higher for Ile aux Aigrettes, even allowing for some of the Ile aux Aigrettes shrews to have been born during the course of the six-month trapping programme.

Experiments to improve trapping efficiency

Captive bait trials

The stomachs of all the shrews caught on Ile aux Aigrettes and Ile de la Passe were removed at post-mortem; the vast majority were empty. In addition, bait placed in the traps was rarely touched. Throughout the trapping programmes on both islands we had not found an effective bait or attractant that unequivocally improved trap success. Therefore we set up a series of captive bait trials in July 2000, aimed at testing the attractive qualities of the baits we had used so far as well as series of novel substances. Bait trials were carried out in a 2 m diameter enclosure, consisting of aluminium sheeting 0.6 m high, situated in a patch of open ground. Experimental baits were placed in shallow containers at regular intervals around the inside perimeter of the enclosure. The 82 shrews used in this part of the project were all individually housed overnight without food, but provided with sufficient water and bedding material. Shrews were introduced into the centre of the enclosure, next to a dish of water and watched for 10 minutes. Anything they ate during the course of the trial was recorded. Eight experimental baits were used in each trial; six baits were used in all 82 trials while the remaining two were 'wild card' baits, changed approximately every 10 trials. This was done in order to test as many substances as possible.

Table 2 Summary of differences between the Ile aux Aigrettes and Ile de la Passe projects.

	Ile aux Aigrettes	Ile de la Passe
Duration (days)	204	20
No. of trap nights	97822	4800
Area (hectares)	25	2
No. of shrews	759	40
Shrews per hectare	29.2	20
Traps per hectare	64	116

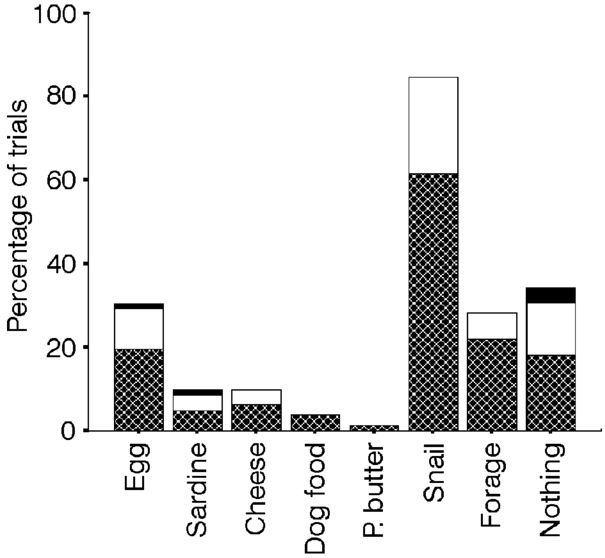

Fig. 3 Relative successes of different bait types, showing contribution made by males (cross-hatched), females (white), and animals of unknown sex (black).

Figure 3 shows the results of the bait trial. Forty-two shrews (52.1%) ate one or more of the baits, 23 (28.0%) ate items found on the floor of the enclosure (some individuals did both), while 28 animals (34.1%) ate nothing. The sex ratio of shrews in this part of the study was strongly skewed, over 2:1 males to females (52 males, 25 females, five unknowns). Figure 3 also shows the contribution males and females made to each result. χ^2 tests showed none of these results to be different from those expected by chance with respect to gender. The results suggest the shrews show a preference for familiar foods (e.g. snail, egg, and foraged items), which could be a profitable line of future bait research. On trial 70, minced snail (*Achatina* spp.) was introduced as one of the wild card baits and proved very successful, being eaten in 11 of the 13 trials in which it was offered. This result was surprising, since captive musk shrews in Guam starved to death rather than eat *Achatina*, when housed with live specimens (Peterson 1956). However, it was an excellent result from the point of view of Mauritian conservation, since *Achatina* are also introduced. They are highly abundant across both mainland Mauritius and Ile aux Aigrettes, and could provide a valuable source of bait, either in a live trapping programme, or as a carrier for a suitable chemical agent. Musk shrews on Ile aux Aigrettes were observed attacking and eating live whole snails, as well as readily taking minced dead snails.

Captive trap trials

Captive trap trials were carried out in July 2000 on 52 shrews to see how they responded to baited and unbaited traps, and to see if foods that showed promise in the bait trial were effective at attracting animals into traps. Eight Trip traps were set around the perimeter of the enclosure, between the dishes of food. Four were baited (with egg, cheese, dog food, or sardines), and four left unbaited. The number of baited and unbaited traps entered by each shrew in a 10 minute period was recorded, along with whether the shrew tripped the trap mechanism or not.

The results are shown in Table 3. Shrews actually entered unbaited traps more frequently than those containing bait, although the difference was not significant ($\chi^2 = 2.73$, df = 1, $p>0.05$). More disturbing was the fact that on one third of occasions, shrews entered and left a Trip trap without causing the mechanism to fire. The most important finding was that if a shrew actually ate the bait present inside the trap (n = 17, 21.8% of trapping events involving baited traps), the trap tripped in every case. This appeared to be due to the shrews moving further into the trap if they ate the bait, which was placed at the back of the trap. So, at least in the case of Trip traps, their performance can be enhanced if they contain palatable bait.

Field bait trials

The captive bait trials had identified some potential baits, but the trap trials suggested this might not be enough to improve trap success. Consequently, in July 2000 we set up a field trial on Ile aux Aigrettes of some of these baits to see if they made any difference to capture rates. A grid of 144 Trip traps was set out using the original 12.5 x 12.5 m grid system in an area of relatively mature ebony forest, the largest section of continuous habitat available. One-third of the traps were baited with boiled egg, the most successful bait known at the time (this trial was set up before the attractive properties of minced snail were discovered), one third with sultanas, which had a zero success rate in the captive bait trials, and the remaining third of the traps were left unbaited. Traps of each type were alternated throughout the grid. The traps were checked every day for seven days (1008 trap nights), and 200 shrews were caught.

The traps baited with egg actually caught fewest shrews; 58 individuals, compared with 78 in sultana-baited traps and 70 in unbaited traps. However, these differences were not significant ($\chi^2 = 5.32$, df = 2, $p>0.05$). Shrews caught in the early stages of trapping programmes are likely to be the most inquisitive individuals, who enter traps out of curiosity rather than in response to the presence of bait. Had we continued this experimental trial for longer, we would have started to catch shrews moving in from untrapped areas (Pilgrim 1996). However, with a larger trapping grid it is possible that the different bait substances may have had an effect on the long-term trapping rate. This study underlines the fact that a bait which proves successful in a captive trial will not necessarily be successful under field conditions.

Table 3 Summary of trapping events (captive trap trials).

	Baited	Unbaited	Total
Tripped	47	63	110
Untripped	31	24	55
Total	78	87	165

DISCUSSION

Recommendations

Trap placement

The results of the captive trap trials, which showed no significant difference in capture rate between baited and unbaited traps, suggest that shrews enter traps out of curiosity rather than in response to the presence of bait. This idea is also supported by the results of the field trial. So, at least at the beginning of trapping programmes, trap placement appears to be more important than the presence or type of bait. Traps must be placed where the animals are most likely to go – along the edges of buildings, rocks, tree roots and paths.

Duration of trapping

Traps need to be left down for long periods, and ideally the whole area should be trapped simultaneously. Some animals do not go into traps for months, and these are the animals we need to target. The failure of the Ile aux Aigrettes removal programme was almost certainly due to missing a very small number of shrews. Setting traps across the whole of the island to be cleared should also increase the chance of trapping stragglers. Because of the extended time period needed, it is also important to trap when the shrews are not breeding. On Ile aux Aigrettes pregnant and/or lactating females were only found between November and April.

Trap type

The results of the captive trap trials show that in one third of cases shrews entered and left Trip traps without being captured. Longworth small mammal traps, used in the first four trapping sweeps of the island, were far less likely than Trip traps to be found tripped without captures - an average of 1.1% of traps per 100 trap nights for Longworths, as opposed to 20.6% for Trip traps. However, Longworth traps are also far more expensive (about £35 each, as opposed to £1 each for Trip traps), and were only on loan to the project. Further research is needed into alternative trap types to see if a cheap, effective alternative can be found. In the meantime, however, Trip traps may have a useful role to play in future eradication attempts in combination with other more effective 'mopping-up' techniques. These may include poisoning (in the event of a suitable agent being found) and possibly the use of specially trained dogs.

Bait type

It remains to be seen if all shrews enter traps solely out of curiosity, or if good bait can eventually encourage the more cautious shrews to enter traps. At the moment, *Achatina* snail seems the most promising candidate. As mentioned above, *Achatina* snails are also introduced to Mauritius, and are also scheduled for eradication from Ile aux Aigrettes. If further bait trials confirm their effectiveness as bait for trapping shrews, it may be wise to combine the eradication plans for these two species. One possibility may be to reduce the population of *Achatina* through handpicking and/or poisoning and then to start trapping shrews using untainted *Achatina* bait when snail numbers are low. However, it is probably sensible to use a variety of different baits during any future trapping programme in order to appeal to as many shrews as possible. Another possibility is using the shrews' own natural scents as attractants, as this species is highly dependent on its sense of smell. Animals in breeding condition have conspicuous scent glands on their flanks, responsible for their characteristic musky odour. Both sexes possess these glands (Dryden and Conaway 1967) and use them for scent marking and communication (Balakrishnan and Alexander 1980). Traps containing adult male shrews could often be detected while still several metres away due to the pungent odour they produced. It may be possible to use the shrews' flank glands as a chemical attractant in traps.

One finding from the trapping on Ile aux Aigrettes was that some shrews will enter traps on the first night, while others will avoid them for months on end despite the high density of traps. It is equally likely that not all shrews respond in the same way to all bait types. In eradication efforts we must strive to target every individual shrew. No single bait type or trap type is likely to appeal to all shrews and future eradication attempts must bear this in mind.

Potential for re-invasion

At present there are no measures in place on either island to prevent new musk shrews becoming established. Limited experiments by Morris and Morris (1991) suggest that shrews are unlikely to reach Ile aux Aigrettes by natural means, due to the strength and direction of the current between the island and mainland. The likelihood of their reaching Ile de la Passe is even more remote given its considerable distance from the mainland. The most likely route for shrews re-invading either island is via shipment of bulky supplies, especially camping equipment, building supplies, and large amounts of food. Ile de la Passe is a popular spot for picnicking Mauritians, especially at weekends and public holidays in summer. During the three weeks of the Ile de la Passe eradication project approximately 50 people visited the island. Some people bring bags of firewood, as well as large quantities of food and camping equipment, all possible hiding places for stowaway shrews. The absence of shrews on nearby Ilot Vacoas is probably due in part to the fact that it makes a less attractive picnic site, being small, flat, and difficult to land on. Continued monitoring of Ile de la Passe is needed to ensure the island remains clear of shrews. If they are found to have re-invaded, they should be removed again as soon as possible. MWF have the equipment and staff to do this at short notice and the island could probably be cleared again by two people in two weeks.

Ile aux Aigrettes is a different situation. If shrews were eradicated in the future it will represent a huge investment of time and resources. Equipment, food supplies, and building materials must be thoroughly checked for the presence of shrews before being brought to the island. It would also be sensible to have a *cordon sanitaire* of traps and/or suitable poison around the jetty and nearby Warden's house. The permanent presence of MWF staff on the island means that re-invaders would hopefully be found quickly before they had time to breed out of control.

Coordinating multi-species eradications

The timing of eradication projects on islands with multiple invasive species needs careful consideration. The shrews on Ile aux Aigrettes were seldom seen prior to the eradication of black rats (*R. rattus*), and it was only after the removal of this competitor that they multiplied to pest proportions. Similar findings have been reported from areas of mainland Mauritius subject to rat control, where shrew numbers increased in inverse proportion to rat numbers (D. Hall pers. comm.). We therefore recommend that every effort is made to ascertain the presence of shrews on islands where rat eradications are planned. This would allow the shrews to be specifically targeted in either a prior or parallel eradication attempt.

The future of trapping for island shrew eradications

On small or topographically simple islands

We appear to have successfully cleared shrews from one small island, showing that the technique can work. However, the chance of failure increases dramatically as the number of shrews increases; obviously, the more shrews there are, the greater the chance of missing one or more animals. On very small or topographically simple islands, trapping may be sufficient in itself to eradicate a population of musk shrews, as appears to have been the case on Ile de la Passe.

Where cheap, dedicated manpower is readily available

The labour requirements and associated costs will also increase as the work area increases, and in many situations will be prohibitive. The failure of the Ile aux Aigrettes project was due in part to not having sufficient traps and human resources in place on the island to react quickly enough to the recurrence of shrews. Trapping programmes of this scale need large numbers of staff who are available for the duration of the programme.

Where the use of poison is constrained

Poison bait trials are required as a matter of urgency to see if an appropriate chemical control method can be identified. Finding an effective poison could revolutionise musk shrew control. However, trapping could still play an important role on islands like Ile aux Aigrettes, where the presence of endangered species constrains the use of large quantities of poison, or where the risk of secondary poisoning through scavenging of carrion is unacceptably high.

In combination with other methods

The future of shrew eradication is likely to rely on a combination of methods, perhaps incorporating trapping and poisoning. Trapping has been shown to be a highly-effective way of dramatically reducing populations of shrews relatively quickly – 75% of shrews caught on Ile aux Aigrettes were caught within eight days, and over 90% within 30 days. On larger islands, trapping may have a role to play as an efficient way of quickly reducing musk shrew numbers locally, but other methods may prove more effective at catching remnant individuals. One possibility may be tracking with dogs specifically trained on the scent of the target species. This method has been used extensively in New Zealand to target the last individuals in eradications of possums (Brown and Sherley 2002) and wallabies (Mowbray 2002), where the populations had already been knocked down through the use of poison. It may be possible to adapt the method to track shrews, using dogs to locate individuals surviving any future large scale trapping or poisoning programmes. However, the use of dogs with this species, which is not naturally preyed upon by canids, is as yet untested and may prove impractical.

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Appendix 1 Summary statistics for shrews caught on Ile aux Aigrettes and Ile de la Passe.

Sample	Sex ratio (M : F)	No. of shrews	Weight (g)		Head + body length (mm)	
			Mean	Range	Mean	Range
Ile aux Aigrettes						
(Removal)	0.72	759*	18.1	10-48	100.2	75-131
Males		317	21.2	12-48	105.1	84-131
Females		441	15.9	10-28	96.7	75-117
Ile de la Passe						
(Removal)	1.22	40	22.0	10-42	105.8	76-129
Males		22	24.7	10-42	108.7	76-129
Female		18	18.7	12-26	102.2	90-116
Ile aux Aigrettes						
(Captive bait trial)	2.04	82**	23.7	15.46	105.3	85-139
Males		51	25.9	15-46	108.7	89-139
Females		25	19.6	15-27	98.8	85-115
Ile aux Aigrettes						
(Field bait trial)	1.4	200#	22.7	14-36	103.7	78-126
Males		112	25.2	16-36	107.2	89-126
Females		80	19.2	14-27	98.8	78-115

* Includes one shrew of unknown sex
** Includes five shrews of unknown sex
\# Includes eight shrews of unknown sex

Eradication of Norway rats (*Rattus norvegicus*) and house mouse (*Mus musculus*) from Browns Island (Motukorea), Hauraki Gulf, New Zealand

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Abstract Browns Island (60 ha) is located within the Waitemata Harbour, Auckland, New Zealand. Mice (*Mus musculus*) were on this island for an unknown period. Norway rats (*Rattus norvegicus*) were first recorded in the late 1980s when their burrows were observed to be damaging archaeological sites. An eradication operation was organised using donated materials and helicopter services. A single application of Wanganui No. 7 bait loaded with bromadiolone at 20ppm was applied by helicopter at a nominal rate of 10 kg/ha in September 1995. One mouse was trapped 19 days after the poison drop but there has been no sign of rodents since. Bait stations placed to intercept possible new arrivals are also used for ongoing monitoring.

Keywords Bromadiolone, historic sites.

INTRODUCTION

Browns Island, 60 ha, is located within the Waitemata Harbour, Auckland, New Zealand, and is separated from the mainland, which has rats, and rat infested islands, by distances of greater than 600 metres at low tide. It is a Recreation Reserve owned by Auckland City Council but managed by the Department of Conservation.

This island has a long history of human occupation with Maori and other historic sites covering much of the land area. This is a highly-ranked site for the conservation of these historic values.

The vegetation today is mainly introduced grasses. A canopy of pohutukawa (*Metrosideros excelsa*) over mainly introduced shrubs exists in limited coastal cliff areas around the north-eastern quarter (Fig. 1) and comprises less than 1% of the island area. There is a lesser area of *Cupressus macrocarpa* and scattered trees of other introduced species. Apart from fenced off coastal cliffs the island has been grazed for nearly 150 years. This is a lowly-ranked site for the conservation of flora and fauna apart from the population of New Zealand dotterel (*Charadrius obscurus*) which utilise the beaches.

The date of mouse (*Mus musculus*) introduction is not known. Rabbits (*Oryctolagus cuniculus*) were introduced in about 1975 and were eradicated between 1985 and 1991 (Veitch 1995). Mustelid sign, probably stoat (*Mustela erminea*), was observed in August 1995. Norway rats (*Rattus norvegicus*) were first recorded in the late 1980s when their burrows were observed to be damaging archaeological sites (Robert Brassey pers. comm.).

The possible impact of this operation on non-target species was considered but no populations were identified as possibly at risk. No action was considered for management of the mustelids as this island is well within their swimming range from the mainland.

The objective of this operation was to remove rodents from Browns Island and thus stop the damage they were causing on archaeological sites.

METHODS

The eradication operation was organised using donated materials and helicopter services. On 13 September 1995, the eradication was initiated with bait loaded into a helicopter bait spreader bucket at North Head (Fig. 1). Wanganui No 7, a 2 gram green dyed pollard pellet containing 20 ppm bromadiolone, was applied at a nominal

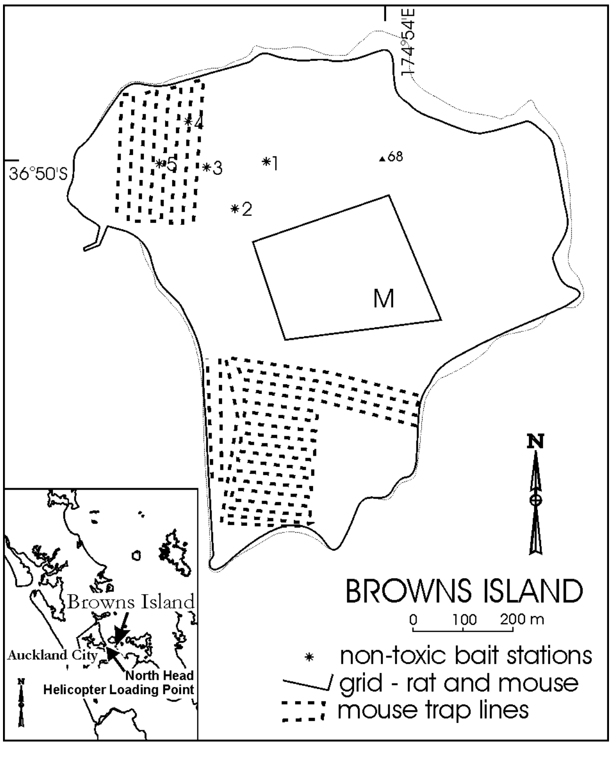

Fig. 1 Browns Island showing post-drop monitoring trap lines and bait stations.

Pages 350-352 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Table 1 Some published LD_{50} values (mg/kg) for brodifacoum and bromadiolone (shown with levels of accuracy and ranges as published). Where the rat or mouse was identified to species level these were chosen over generic groups. Where "wild" was specified this was chosen over captive or unspecified strains.

Brodifacoum		Bromadiolone		
Mouse	Rat	Mouse	Rat	Reference
0.40 (0.30-0.63)	0.22	0.86 - 1.75	0.57-0.65	Hone & Mulligan 1982
0.4	0.27	0.99	0.65	Haydock & Eason 1997
0.4	0.24	1.75	1.125	Eason 1991

rate of 10 kg/ha. This was used subject to an experimental use permit.

The helicopter was not fitted with a Differential Global Positioning System (DGPS) to aid the accuracy of bait placement. Instead, a colour aerial photograph was marked with flight lines spaced to ensure adequate overlap of bait spread. The small size of the island and presence of suitable landmarks such as trees, stone walls and fence lines made this a reasonable practice.

No attempt was made on the ground to check for gaps in the bait spread as the dense grass sward made it impractical to do so. A period of more than three days fine weather followed the bait drop. Random searches to assess the quantity of bait remaining were made from 26 September to 5 October and on 10 October.

From 26 September to 5 October 121 Ezeset mouse snap traps were set for 968 trap nights and 25 Ezeset rat snap traps were set for 200 trap nights on a grid (Fig. 1). Five stations with 3 cm cubes of cheese fixed on wires under tin covers were also set out for this period. This non-toxic bait trial was chosen following bait preference trials on this island earlier in the year (Weihong *et al.* 1999). From 30 October to 9 November, 386 mouse snap traps were operated along lines for 3860 trap nights (Fig. 1).

For ongoing protection of the island, and as a monitoring mechanism, 50 Rentokil bait stations containing Rid Rat wax block baits were placed around the island. The bait in these stations is replaced at six-monthly intervals and checked periodically for rodent sign.

RESULTS

One mouse was trapped on the grid 19 days after the poison drop (shown as 'M' on Fig. 1). Mouse sign in the form of chewed baits was observed at two of the five bait stations over the latter part of the period they were set out; up to 21 days after the bait drop. No rats or mice were caught on the mouse trap-lines. There has been no sign of rats or mice since 5 October 1995.

The random searches for remaining bait which began 13 days after the drop revealed only a few small pieces of bait, each less than 0.25 g.

DISCUSSION

Bromadiolone at 20 ppm in a single aerial drop of 2 g pellets was successful in the eradication of rats and mice from Browns Island.

Brodifacoum has been the toxin of choice for rodent eradications (e.g. Brown 1993, Taylor and Thomas 1993), as it is very highly toxic to rodents, and when compared with other less toxic rodenticides, less active ingredient is required to kill the target species. We used bromadiolone because it was donated, thus reducing our costs.

The LD_{50} data for brodifacoum and bromadiolone and the two rodent species involved in this operation vary between studies (Table 1). If worst case scenario data are used then both rats and mice may need to eat more than four times the quantity of bromadiolone loaded bait compared to brodifacoum loaded bait (Table 2). In the Browns Is-

Table 2 A comparison of the quantities of brodifacoum and bromadiolone that may need to be consumed by mice and Norway rats to meet the LD_{50} level. The highest LD_{50} levels from Table 1 have been used. Rodent weights are from King (1990).

	Brodifacoum	Bromadiolone
Mouse LD_{50} (mg/kg)	0.40	1.75
Norway rat LD_{50}	0.27	1.13
mg of toxin to meet LD_{50} level		
Mouse if 24 g	0.01	0.04
Norway rat if 260 g	0.07	0.29
Grams of bait loaded at 20 ppm		
Mouse if 24 g	0.48	2.10
Norway rat if 260 g	3.51	14.69
Number of feeding days bait loaded at 20 ppm		
Mouse[1]	0.16	0.70
Norway rat[2]	0.14	0.56

[1] Crowcroft (1996) – a mouse consumes 3-4g of food daily – 3g level used here.
[2] Leslie and Ranson (1954) – Norway rats eat about 10% of their body weight daily = 26g/day.

land situation it appears that the Norway rats rapidly cached the baits which was then available to the mice after the rats had died. Some individual rodents may need to ingest three times the LD_{50} level to obtain a lethal dose. Thus a Norway rat on Browns Island may have needed to eat only bait for 1½ days and a mouse may have needed to eat only bait for more than two days to receive lethal doses.

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Eradication of Norway rats (*Rattus norvegicus*) and house mouse (*Mus musculus*) from Motuihe Island, New Zealand

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Abstract Motuihe Island is located within the Waitemata Harbour, Auckland, New Zealand. Mice (*Mus musculus*) and Norway rats (*Rattus norvegicus*) were introduced to this island at an unknown date. Rabbits (*Oryctolagus cuniculus*) and feral house cats (*Felis catus*) are present. Most of the island is a pastoral farm. Past anecdotal records of rats and mice suggest that there have been significant changes in their abundance from year to year. Two aerial applications of Talon 7-20, a cereal-based anticoagulant rodenticide containing brodifacoum, were made in 1997 with the intention of eradicating both the rats and the mice. Trapping for rats and mice in 1999 and 2000 failed to detect the presence of either species and the project is deemed to have achieved eradication.

Keywords Brodifacoum.

INTRODUCTION

Motuihe Island, 179 ha, is located within the Waitemata Harbour, Auckland, New Zealand (Fig. 1), and is separated from the mainland and rodent infested islands by distances of greater than 1.05 km at low tide. This island has a long history of human use, including numerous Maori sites. Since about 1840 it has been a pastoral farm. It has also been used as a quarantine station, prisoner-of-war camp, and naval training base. It is now a Recreation Reserve managed by the Department of Conservation with about 80% of the land area grazed by sheep and cattle (DOC 1995).

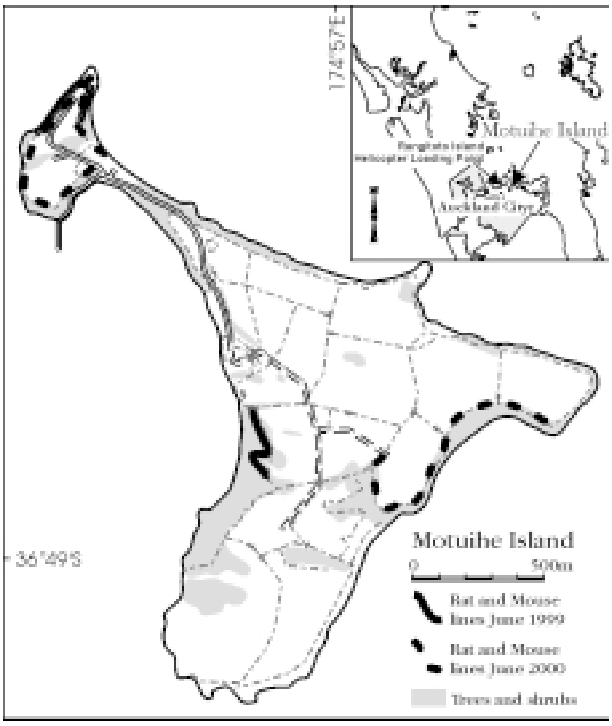

Fig. 1 Motuihe Island showing vegetation and the location of the traplines used in June 1999 and June 2000.

Most of the shoreline is sandy beach with much of it backed by vertical cliffs up to 30 m high. The farmed land is rolling and reaches a maximum altitude of 63 m. The 20% of the island that is not grazed pasture has stands of native and non-native trees and is heavily grazed by rabbits (*Oryctolagus cuniculus*).

The holders of a concession to operate the small shop, manage camping and day visitor facilities, farm the land and let one house, live on the island. Visitor numbers are not recorded but on a good summer day 600 boats, varying in size from kayaks to 20 m in length, may be hauled ashore or anchored nearby.

There is no definite information on when the mice (*Mus musculus*), Norway rats (*Rattus norvegicus*), cats (*Felis catus*) or rabbits arrived on the island and there are no records of rodent abundance prior to the 1995 study by Stubbs (1996). Cats were recorded as being eradicated in 1981 (Veitch and Bell 1990) but have since returned. Mustelids have never been recorded.

Rats were abundant from time to time prior to 1987, but mice were not seen (John Allen pers. comm.; Chris Roberts pers. comm.). Poisoning operations for rabbits in 1988 may have eradicated Norway rats (Dowding *et al*. 1999). Between March and December 1995, Stubbs (1996) set rat and mouse traps at six-weekly intervals for a total of 1260 trap nights and caught 212 mice, but no rats. The capture rate varied from four to 45 mice/100 trap nights. In February 1997, I operated 30 Ezeset rat traps and 30 Ezeset mouse traps for three nights and caught one Norway rat.

Secondary objectives of this bait drop, not part of this paper, were the reduction of rabbit numbers prior to their eventual eradication and the assessment of toxin transfer from rat baits to cats, by consumption of prey that had eaten toxic baits (Dowding *et al.* 1999), prior to their eventual eradication using other methods. This work was not completed due to a withdrawal of funding.

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The possible impact of this operation on non-target species was considered, based on reported and personal experience of previous rodent eradication operations, but, while individuals were expected to be affected, no populations were identified as possibly at risk.

METHODS

Using a helicopter, two aerial applications of Talon 7-20, 2 gram baits, were made with the intention of eradicating both the rats and the mice. Talon 7-20 is a cereal-based bait containing brodifacoum at a rate of 20ppm and dyed green to reduce visual attraction to birds. This was used because it was the only bait registered for aerial application on offshore islands. After the operation was completed it was found that this bait also contained bitrex, a bittering agent added to the toxin to reduce the possibility of humans eating the bait. Bitrex has been shown to reduce bait consumption by rats (Kaukeinen and Buckle 1992; Veitch 2002b).

The bait was spread from a dedicated underslung bait bucket which spread the bait over a 120m wide swath. The helicopter was fitted with a Differential Global Positioning System (DGPS). For both drops the DGPS was set so that the helicopter pilot would follow a line spacing of 120 metres.

In previous rodent eradication operations bait had been applied at or about 10 kg/ha and successfully eradicated the target species (e.g. Brown 1997; Veitch 2002a. 2002c). For this operation two rodent species were present, so two bait spreads were considered desirable, with an eight day interval between drops. As two drops were to be made, a lower bait application rate was possible for each drop, with the total bait applied being slightly more than for a single-species operation.

The first drop of bait was carried out on 25 July 1997 with 1450 kg of Talon 7-20 being spread at a nominal rate of 8 kg/ha. The second drop of bait was applied on 4 August, used 800 kg of bait and the bait bucket was set to spread at 4kg/ha. A printout from the DGPS (Fig. 2) was checked for gaps between flight lines after each bait application.

There was no ground verification to monitor the thoroughness of bait coverage over the island. The spread rate was monitored immediately after the first drop by randomly casting a metre-square wire frame to left and right of a casual line of about 600 m walked over grazed pasture between the summit and the woolshed. The baits within each metre square where the frame fell were counted. This test was repeated 50 times.

Between 28 June and 2 July 1999, two trap-lines each with 30 Ezeset rat traps under tin and wire mesh covers and 30 Ezeset mouse traps under wire mesh covers were operated for a total of 480 trap nights (Fig. 1). These traps were set at 30 metre intervals in the order: rat trap under tin cover; mouse trap under mesh cover; rat trap under mesh cover; mouse trap under mesh cover; and so on. The traps were set in rough vegetation under fences or just into the scrub edge beyond the farm stock browse line.

Between 12 and 16 June 2000 two trap-lines of alternating rat and mouse traps set in the same order and similar locations as in 1999, but at 50 m intervals, were operated for a total of 380 trap nights (Fig. 1).

There was no organised or regular searching for non-target species, but birds found dead and suspected of being poisoned were collected for analysis.

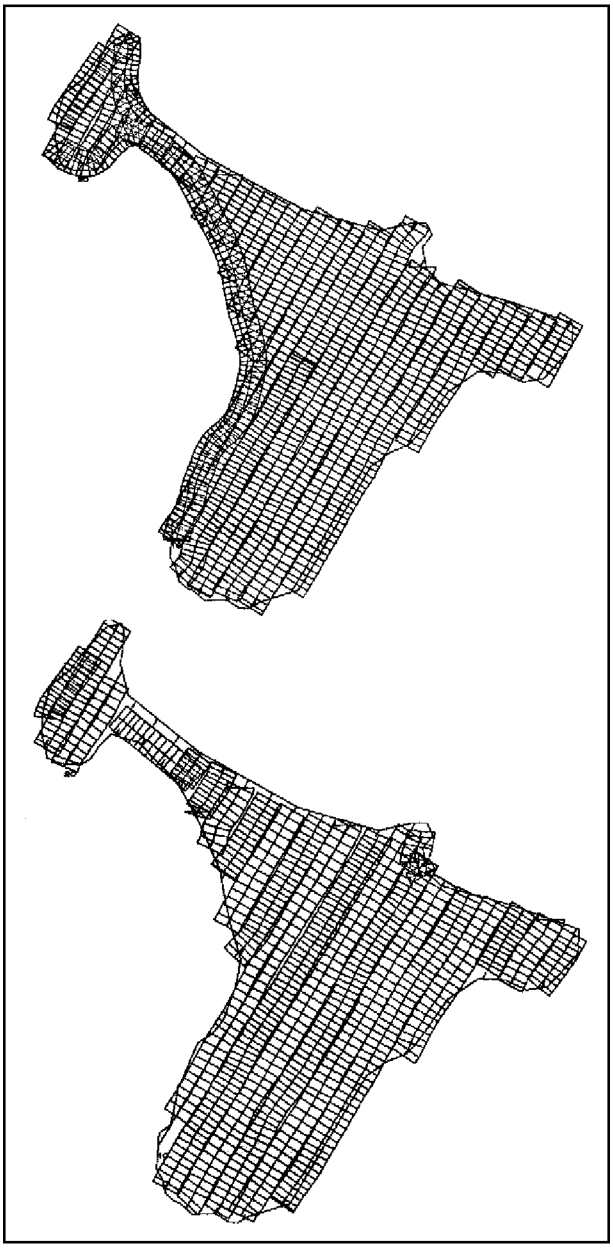

Fig. 2 Upper: The DGPS printout following the first bait drop on 25 July 1997. The line indicating the island outline is not the surveyed outline but is the helicopter track when logging the outline. Lower: The DGPS printout following the second bait drop on 4 August 1997.

RESULTS

The DGPS printout for the first drop (Fig. 2) showed a thorough flight coverage of the island. The weather on Motuihe remained dry over the next three nights. The DGPS data shows that the 1450 kg of bait was spread over 235 hectares (6.18 kg/ha) but there are some areas of overlap and some flight lines extend beyond the island boundary. The test of bait spread using a metre-square frame found a bait spread equal to 7.96 kg/ha.

Analysis of the DGPS data shows that the second drop of bait achieved a spread rate of 3.5 kg/ha over the logged application area. A strong south-east wind was blowing at the time and this shows up in the relative flight speeds of the up-wind and down-wind tracks shown on the DGPS printout (Fig. 2). One day after this drop an absence of bait was noticed on the north-western headland (John Dowding pers. comm.), and a subsequent search failed to find any bait on this headland. This part of the island was the end of the helicopter flight and these observations suggest that the bait bucket was empty before the helicopter reached this point. No bait was available to remedy this problem.

Twenty-nine individuals of 10 bird species found dead following the operation were collected: paradise shelduck (*Tadorna variegata*), mallard (*Anas platyrhynchos*), grey duck (*A. superciliosa*), Australasian harrier (*Circus approximans*), pukeko (*Porphyrio p. melanotus*), southern black-backed gull (*Larus dominicanus*), blackbird (*Turdus merula*), chaffinch (*Fringilla coelebs*), common myna (*Acridotheres tristis*) and Australian magpie (*Gymnorhina tibicen*). Analysis of the livers of these birds showed that all contained brodifacoum (Dowding *et al*. 1999). House sparrows (*Passer domesticus*) and goldfinches (*Carduelis carduelis*) were observed eating bait but none were found dead. Three groups of pukeko that were counted before and after the operation were observed to decline by 49%, but have since increased again, and a paradise shelduck flock declined by 60% (Dowding *et al* 1999). Both pukeko and paradise shelduck have now returned to their pre-bait drop abundance.

Trapping for rats and mice in 1999 and 2000 failed to detect the presence of either rodent species. There has been no other sign which might indicate the presence of rodents. Rats and mice have probably been eradicated.

DISCUSSION

The DGPS data presented here suggests that there are considerable discrepancies between the intended rate of bait spread and the actual rate of bait spread. However, the one ground check recorded a bait spread similar to the intended spread. The helicopter pilot needs to operate two switches at the beginning and end of each flight line: one to switch the bait flow on or off and the other to start or stop the DGPS track. There will be a time lapse between these two actions and this is accentuated when operating on small islands with short bait spreading runs. In this instance if the time lapse averaged one second then the recorded bait spread rate differs from the actual bait spread rate by 8%. On this island, unlike forest covered islands, ground checks of bait spread were possible and more checking should have been done.

The high variability of rodent density, both anecdotal and from the trapping data recorded in this paper, may be due to a number of factors either singly or in combination:

- The rabbit population has frequently been at high levels and their browsing is likely to reduce the food source for rodents.
- There has been intermittent action by island managers to control rabbits using a wide variety of methods which also kill rodents (J. Allan pers. comm., Dowding *et al*. 1999).
- Cats are present and they may slow the rate of increase of a reduced rodent population (Fitzgerald 1990).

This variation in rodent abundance, combined with the low trapping success in February 1997, means that the failure to catch rodents in 1999 and 2000 may not be confirmation that rodents are absent. Their continuing absence from buildings, the compost heap, and rubbish containers suggests that eradication was successful. If a mouse or Norway rat appears again after these five years of absence there is no way of knowing whether the eradication operation failed or it is a new arrival from one of the many boats that visit the island.

ACKNOWLEDGMENTS

I thank Ian McFadden and John Dowding for their assistance with technical advice and information during this operation and staff of the Auckland Field Centre for their assistance during bait spreading.

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Eradication of Pacific rats (*Rattus exulans*) from Fanal Island, New Zealand.

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Abstract Fanal Island (73 ha) is the largest island in the Mokohinau Group, Hauraki Gulf, New Zealand. Pacific rats, or kiore, (*Rattus exulans*) reached these islands between about 1100 and 1800 A.D. Pacific rats were removed from all islands in this group, except Fanal Island, in 1990. An aerial application of Talon 7-20 (containing brodifacoum at 20 ppm) at a nominal rate of 10 kg/ha was made on Fanal Island on 4 August 1997 with the intention of eradicating the rats. Despite heavy rainfall immediately after the poisoning operation, and the fact that baits were not of optimum palatability, the rats were eradicated.

Keywords Bitrex; brodifacoum; aerial baiting.

INTRODUCTION

Fanal Island (Motukino), 73 ha, is the largest and most southerly island in the Mokohinau Group which is at the northern extremity of the Hauraki Gulf, about mid way between Great Barrier Island and the mainland (Fig. 1). Fanal Island is part of Mokohinau Islands Nature Reserve, which is administered by the Department of Conservation under the Reserves Act 1977. It is also a wildlife sanctuary under the Wildlife Act 1953.

Pacific rats, or kiore, (*Rattus exulans*) are presumed to have reached these islands with Maori between about 1100 and 1800 A.D. These islands were also modified by burning of the forest to aid Maori food gathering and Burgess Island, the next largest in the group, was cleared for pastoral farming associated with the establishment of a lighthouse (*c.* 1890) and wartime defence operations. The impacts of Pacific rats on these ecosystems are not known, but from circumstantial evidence and studies at other locations they were presumed to be detrimental to natural processes (Holdaway 1989; Atkinson and Moller 1990). Pacific rats were removed from all islands in this group, except Fanal Island, by the Department of Conservation in 1990 (McFadden and Greene 1994).

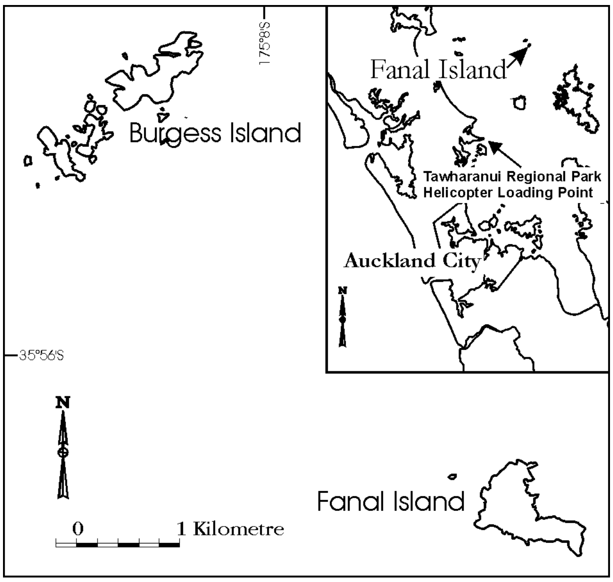

Fig. 1 Fanal Island is part of the Mokohinau Group situated at the northern extremity of the Hauraki Gulf.

Under the Regulations Act 1936 and the Grey-faced Petrel (Northern Muttonbird) Notice 1979, Ngati Wai of Aotea (Great Barrier) have muttonbirding privileges on all islands of the Mokohinau Group. They approved the removal of Pacific rats from the northern Mokohinau Group, and they initiated the proposal to remove Pacific rats from Fanal Island in May 1995.

Fanal Island is surrounded by steep cliffs and has a gently sloping forested summit plateau, which rises to 134 m near the northern cliffs. There are small seasonal streams in the three main valleys which drain to the south-west. The vegetation is coastal forest and scrub, much of which is regenerating following fires in earlier times (Esler 1978; Wright 1980a, 1980b; de Lange *et al.* 1994).

Most of the expected array of forest birds was present on Fanal but saddlebacks (*Philesturnus carunculatus*) were absent, presumably as a result of previous forest destruction. Attempts to re-introduce saddlebacks in 1968 and 1985 failed to establish a breeding population although individuals did survive up to 15 years after liberation.

During preparations for this operation no populations of fauna were identified which might be detrimentally affected by the proposed aerial bait distribution to eradicate the rats.

METHODS

Bait spread

On 4 August 1997 an aerial application of Talon 7-20 was applied to Fanal Island at a nominal rate of 10 kg/ha. Talon 7-20 is a pollard bait containing brodifacoum at 20 ppm and the bittering agent bitrex (denatonium benzoate).

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The operation was run from Tawharanui Regional Park (Fig. 1), with permission of the Auckland Regional Council Parks managers. A Differential Global Positioning System (DGPS) base station was established on the hilltop east of Park Headquarters at Tawharanui Regional Park. Marine Helicopters Ltd Llama helicopter took one underslung bait spreader bucket loaded with 825 kg of Talon 7-20 bait to Fanal Island. A strong south-east wind was blowing. The island boundaries were logged into the DGPS and the bait was then spread.

The DGPS was set to show navigation lines for the helicopter at 60 metre intervals. The bucket being used was spreading bait over a 120 m swath at a rate of 4 kg/ha. Thus we expected 8 kg/ha to be spread in most areas, 4 kg/ha where there had been only one pass, and an average of 10 kg/ha over the flight area.

Monitoring

Unpredicted heavy rain fell between 5pm on 5 August and 1am on 7 August (Fig. 2). The total fall recorded by the automatic weather station on Burgess Island was 106 mm with hourly rates averaging 3.4 mm per hour and reaching a maximum of 9 mm per hour. There were no visits to the island to monitor possible acceleration of bait decay due to this rain.

During a brief visit to the island in mid-1998 rat traps were set for one night (I. McFadden pers. comm.). During a visit in May 1999 snap traps were set for 400 trap nights and extensive searches were made for fresh rat sign (P. Todd pers. comm.). Further searches for rat sign were made, but no traps were set, during a visit in May 2000 (G. Wilson pers. comm.).

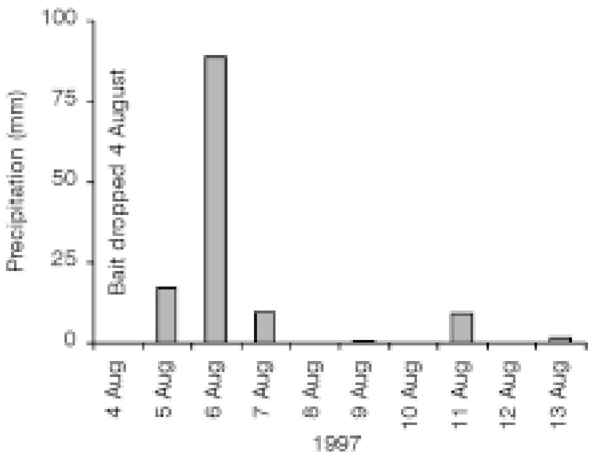

Fig. 2 Rainfall recorded at the automatic weather station on Burgess Island. This is for the 24 hours of each day, not to 0900 hours as at manual weather stations.

RESULTS

No rats or rat sign was detected during the three visits in 1998, 1999, and 2000 (I. McFadden pers. comm.; P. Todd pers. comm.; G. Wilson pers. comm.). It is believed that Pacific rats have been eradicated from Fanal Island despite significant rainfall over the second and third nights after the bait was spread and the presence of bitrex in the bait.

DISCUSSION

In this situation, where ground checking and a second flight to spread more bait was not possible, the increased flying time to give closer flight lines proved an acceptable alternative. The overlapping flight lines ensured that there was a complete bait coverage of the island with a rate of 4 kg/ha being the lowest likely spread.

Bitrex, a bittering substance added to reduce accidental human consumption of rat bait, has been shown to reduce the consumption of bait by captive Pacific rats (Appendix 1). We were not aware of the presence of bitrex in the Talon 7-20 formulation until after the operation but it did not affect the final outcome of the operation.

The period of heavy rainfall was not predicted in weather forecasts or seen by weather observers until late on 4 August (R. McDavitt pers. comm.). It is unfortunate that monitoring of bait decay was not possible and it is not known whether bait survived this rain or whether the rats all consumed lethal doses within the first two nights.

The success of this operation, despite adverse weather conditions and unattractive bait, is evidence of the effectiveness of aerial application of brodifacoum baits for rat eradication from islands. Nevertheless, it is prudent that future operations be managed with such variables in their favour.

ACKNOWLEDGMENTS

I thank Bill Simmons and Animal Control Products for the bait supplied for trials and Ian McFadden for his technical advice and reports on the follow-up monitoring.

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Appendix 1 A comparison of baits with and without bitrex.

In May 1996 baits with and without bitrex were tested on wild caught Pacific rats (*Rattus exulans*) on Little Barrier Island as part of a larger test of bait flavours.

Bitrex (denatonium benzoate) may be added to brodifacoum at the point of manufacture. The brodifacoum in the samples compared came from two different sources; one contained bitrex and the other did not. The quantity of bitrex in the toxin is not known. In all other respects the baits were the same, being Wanganui No. 7 formulation dyed green and pressed into 2 gram pellets. The brodifacoum was loaded at 20 ppm. The bait with bitrex is now the registered formulation Talon 7-20 and the bait without bitrex is now the registered formulation Pestoff Rodent Bait 20R.

Fifteen rats were used to compare consumption of bait with and without bitrex. Each rat was kept in a separate cage which contained a shelter and water supply. Each rat was offered two shallow containers: containing *c.* 40 grams of the bait with bitrex; the other containing *c*. 40 grams of the bait with no bitrex. The bait was reweighed next morning. Results are shown in Table 1.

Twelve of the 15 rats chose to eat significantly less of the bait that contained bitrex. The differences in consumption were highly significant P= 0.0004 (TTEST in Microsoft Excel). The consumption of non-bitrex bait was more than three times greater than the consumption of bait containing bitrex.

Table 1 Amount of bait (grams) with and without bitrex consumed by captive Pacific rats on Little Barrier Island in May 1996.

	Sample size	Average consumption (g) with range	Standard Deviation
With bitrex	15	2.9 (0-17)	4.82
Without bitrex	15	9.7 (0-17)	4.57

Eradication of Pacific rats (*Rattus exulans*) from Tiritiri Matangi Island, Hauraki Gulf, New Zealand

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Abstract Tiritiri Matangi Island (220 ha) is 25 km north of Auckland City in the Hauraki Gulf. Pacific rats, or kiore, (*Rattus exulans*) were probably introduced to this island between 1100 and 1800 A.D. The impacts of Pacific rats on this ecosystem are not known, but from studies at other locations they were presumed to be detrimental to natural processes. Until 1984 the natural ecosystem was also degraded by removal of forest cover and pastoral farming. Restoration of the natural environment on this island began in 1984. Some 300,000 native trees have been planted and nine species of native bird translocated to the island. Pacific rats were removed in September 1993 with use of an aerial application of Talon 20P rodent bait. Specific actions were taken during this operation to protect identified potential non-target species where the population was considered at risk. Monitoring methods following the operation are described. There has been no rat sign found since the aerial operation.

Keywords Eradication; brodifacoum; Pacific rat, *Rattus exulans*; non-target impacts.

INTRODUCTION

Tiritiri Matangi is a low-lying island of 220 ha lying 4 km off the Whangaparaoa Peninsula and 25 km north of Auckland City in the Hauraki Gulf, New Zealand. It is a Scientific Reserve under the Reserves Act 1977 and is open to public visitation. In the year ended June 1994, more than 16,000 people visited Tiritiri Matangi (B. Walter pers. comm.).

Maori occupied Tiritiri Matangi prior to the arrival of Europeans in New Zealand and, from at least 1841, it was grazed by domestic animals. A lighthouse was established on the south-eastern end of the island in 1865. The Crown withdrew the grazing lease in 1971 and management of the island was then taken up by the Hauraki Gulf Maritime Park Board. At that time it was proposed that, apart from the Lighthouse Reserve area, native vegetation be allowed to regenerate naturally.

Cats (*Felis catus*), rabbits (*Oryctolagus cuniculus*), and goats (*Capra hircus*) have been reported as being established feral populations on Tiritiri Matangi which were subsequently removed. Cats were probably never established as a feral population and the occurence referred to by the Dept. of Lands and Survey (1982), and later quoted by Moller and Craig (1987), related to domestic cats owned by a lighthouse keeper (A. Wright pers. comm.). Rabbits which were at one time plentiful had disappeared by 1908 (Dept. of Lands and Survey 1982). The goat population was small and was removed by the lighthouse keepers. This work was under way in 1961 (A. Wright pers. comm.) and no goats were present in 1971 (R. Walter pers. comm.).

In 1979 a programme of planting to enhance regeneration was proposed, with a plan calling for the planting of most of the island while leaving selected areas to regenerate naturally (Fig. 1). Since 1984 more than 300,000 native trees have been planted increasing the proportion of non-grassland vegetation from 6% to 60% of the island's area (Galbraith and Hayson 1995), but planting has now ceased.

Between 1973 and 1998 nine species of native bird were introduced to this island. These introductions have been for two purposes: restoration of the island ecosystem (species marked [R]) and providing a refuge for threatened species (species marked [T]). The species are: red-crowned

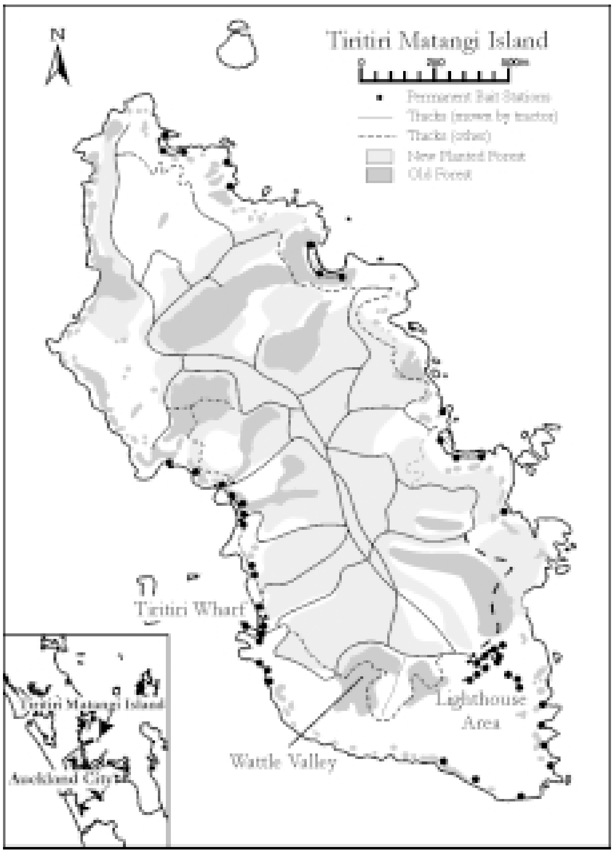

Fig. 1 Tiritiri Matangi Island showing the wharf, tracks, buildings and the placement of permanent bait stations.

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parakeet [R] (kakariki) (*Cyanoramphus novaezelandiae*) 1973; North Island saddleback [R] (tieke) (*Philesturnus carunculatus*) 1984; brown teal [T] (pateke) (*Anas aucklandica chlorotis*) 1987; whitehead [R] (popokatea) (*Mohoua albicilla*) 1989; takahe [T] (*Porphyrio mantelli*) 1991; North Island robin [R] (toutouwai) (*Petroica australis*) 1992; little spotted kiwi [T] (kiwi pukupuku) (*Apteryx owenii*) 1993; stitchbird [T] (hihi) (*Notiomystis cincta*) 1995; and North Island kokako [T] (*Callaeas cinerea*) 1998.

The Pacific rat, or kiore, (*Rattus exulans*) was present on the island at the time of first European records but was removed in the operation during September 1993 described in this paper. The impacts of Pacific rats on this island ecosystem are not known from studies at this location, but other studies (e.g. Whitaker 1973; Atkinson 1978; Craig 1986; Atkinson and Moller 1990; Towns 1991; Holdaway 1999) show that they are detrimental to natural processes.

This eradication was instigated and funded by The Supporters of Tiritiri Matangi Inc. with financial support from the Royal Forest & Bird Protection Society. The operation was managed by Department of Conservation staff from the Auckland Conservancy office and Auckland Area office with technical oversight from the Department of Conservation Island Advisory Group.

METHODS

Poisoning operation

Bait stations and aerial bait spread were both considered for this operation. Aerial bait spread was chosen as it was believed it would require the purchase of less bait, have a greater probability of success on steep coastal areas, achieve eradication over a shorter time period, and cost less overall.

An aerial application of 2.2 tonnes of 2 gram Talon 20P pellets was spread at a nominal rate of 10 kg/ha across Tiritiri Matangi Island on 29 September 1993 with the intention of eradicating the rats. This pollard bait was manufactured to contain brodifacoum at 20ppm but no samples were tested to check the toxin level.

The bait was distributed from a helicopter fitted with a Differential Global Positioning System (DGPS) which was set to show the pilot a line spacing of 100 metres. The boundary of the island was flown so that this would be recorded in the navigation system. The bait was spread from an underslung dedicated bait bucket set to spread bait at 8 kg/ha to a swath width of 120 metres. Thus, when overlaps between flight lines were allowed for, an overall sowing rate of 10 kg/ha was expected.

No ground checks were carried out to check for gaps in bait distribution, as the dense vegetation on the island made it unlikely that this would be successful. As soon as all bait had been spread, a careful check was made of the DGPS record of helicopter travel immediately after the bait drop, before the helicopter left the island.

A statistically-robust system to check bait spread was also not possible, but four checks were made to determine the density of bait spread. Three of these were 50 x 1 m areas on the gravel road. The fourth check was that all bait that fell onto a covered takahe enclosure (20 x 12 m) was collected when the polythene cover was removed.

To monitor bait decay, two plots, each containing 30 baits spread a few centimetres apart, were established on the day of the airdrop. The baits were placed a few centimetres apart on the ground and covered with a 15 mm wire mesh cage to prevent disturbance by rats or birds. One plot was under the canopy of a ngaio (*Myoporum laetum*) tree and the other was in open grassland well away from trees. Ground searches for bait were made on 2 November and 20 November 1993 to assess how much bait remained as potential food for any remaining rats and whether the captive non-target species could be released.

To check for the possible presence of rats after the poison drop, bait stations were set out from 17 to 30 November 1993. Each station contained a Rentokil wax block rodent bait and a piece of toilet soap secured to a board under a tin cover. In total 350 such stations were used at approximately 30 m intervals on the edges of all major tracks on the island.

Reduction of non-target effects

Five bird species were considered to be at risk from primary poisoning as they were known or suspected to take bait. The potential impacts on the pukeko (*Porphyrio melanotus*), North Island robin (*Petroica longipes*) and North Island saddleback (*Philesturnus rufusater*) were not considered likely to be detrimental at the population level although some mortality of individual birds was anticipated. The potential impacts on the brown teal (*Anas aucklandica*) and takahe (*Porphyrio mantelli*) populations were considered to be detrimental at the population level and action was taken to protect these two species during the poisoning operation.

Brown teal had been liberated on the island in 1987 and 1990 but the number present in 1993 was unknown. Between 15 and 21 September 1993 nine brown teal were captured – four by hand netting and five by hunting with a trained dog. These birds were kept in an aviary on the island until 11 November. This aviary was covered with a polythene sheet during the bait drop.

Four takahe were present on the island in 1993: a pair with a two-egg nest and an attendant juvenile from the previous year; and a single male. A 16 x 20 metre enclosure with a one metre high fence was built around the area containing the nest to keep the three takahe within this area. The vegetation in this area varied from open grassland to three metre high trees. Before the poison drop a polythene sheet was drawn over this entire area. For the single male takahe a 20 x 12 m enclosure, containing mostly grassland with a few low shrubs, was constructed. Before the bait drop this takahe was locked in the island's potting shed and a polythene sheet placed, mostly at ground level, over the enclosure. After the drop the polythene sheets were removed. All takahe remained in their enclosures until 3 December, 61 days after the bait drop.

To monitor the impact of this poison drop on non-target species, many people searched randomly for dead birds at irregular intervals over the month following the drop. Birds found in reasonably fresh condition were placed in a freezer for necropsy and residue analysis. The colour-banded populations of robins and saddleback had been monitored since their introduction to the island in 1984 and 1992 respectively. They continued to be monitored throughout the operation. Bird count data gathered by members of the Ornithological Society of New Zealand since 1987 have since been analysed (Graham and Veitch 2002).

For ongoing protection of the island and monitoring for rodent presence, 51 Rentokil rodent bait stations each containing one Rentokil wax block bait containing bromadiolone at 50 ppm were placed at potential landing places (Fig. 1). These have been inspected monthly since 1993 and the bait is refreshed at three to six month intervals.

RESULTS

The checks of the DGPS indicated that bait spread over the island had been thorough.

The bait density checks on the road located 14, 17, and 15 baits in each 50 x 1 m plot. This is equivalent to about 6 kg/ha, but baits may have bounced off this hard surface, thus reducing the count. On the takahe pen cover 180 baits were found, equivalent to about 15 kg/ha, suggesting a double bait spread over this particular area.

An unplanned check of bait abundance in randomly selected areas of grassland and forest five days after the bait drop located two to seven baits per $50m^2$ searched, or about 0.8 to 2.8 kg/ha. This indicates that there was adequate bait for all rats to have had access to bait during the days immediately after the drop.

In the 24 hours up to 0900 on 30 September, the second night after the bait drop, 1.1 mm of rain were recorded on Tiritiri Matangi Island. The next rainfall was 9.7 mm on 13 October. The baits in the bait decay plots remained whole but slowly became blackened with mould. A further unplanned bait search on 2 November revealed no bait in grassland areas and in most forest areas but some was found at the foot of a steep rock face (J. Henry pers. comm.). This was assayed and found to contain 22.1 ppm brodifacoum. No bait was found during the ground search on 20 November.

By 20 November, 52 days after the drop, the pellets in the bait decay monitoring plots were still whole but totally black with mould. A sample of this was found to contain 30.6 ppm brodifacoum. The bait that was spread was presumed to contain 20 ppm but this was not tested so it is not known how much the brodifacoum content of this decayed bait varied from the probability of the fresh bait carrying more than 20 ppm.

Some 302 person hours were spent, mostly by volunteers with varying skill levels, searching for dead birds during the month following the bait drop. The 42 birds found and an assessment of their condition and the probability of brodifacoum poisoning are listed in Table 1.

Table 1 Birds found dead in November 1993 after the aerial poison drop to eradicate Pacific rats from Tiritiri Matangi Island. There was no autopsy of some birds as they were too decayed or had been recorded as found and discarded. Birds which were examined were recorded as poisoned if haemorrhaging consistent with brodifacoum poisoning was found. The assay results are μg/g brodifacoum in each liver sample.

Species	Date	Autopsy	Brodifacoum (µg/g)
Paradise shelduck *Tadorna variegata*	14/10	1 poisoned	
Brown teal	7/10	1 poisoned	
Anas aucklandica	8/10	1 poisoned	
	12/10	1 no autopsy	
Spotless crake *Porzana tabuensis*	12/10	1 no sign	0.04±0.01
Pukeko	5/10-10/10	4 poisoned	
Porphyrio melanotus	6/10-31/10	17 no autopsy	
Red-crowned parakeet *Cyanoramphus novaezelandiae*	23/10	1 see note 1	0.00
Skylark *Alauda arvensis*	14/10	1 no autopsy	
Blackbird *Turdus merula*	16/10-30/10	4 no autopsy	
Chaffinch	7/10	1 poisoned	
Fringilla coelebs	7/10	1 no autopsy	
House sparrow *Passer domesticus*	13/10	1 no autopsy	
Common myna *Acridotheres tristis*	9/10-29/10	4 no autopsy	
Saddleback	15/10	1 poisoned	0.46±0.10
Philesturnus	17/10	1 no autopsy	
carunculatus	24/10	1 no autopsy	

Note 1. This parakeet had haemorrhaging under breast skin and around the heart which could have been consistent with brodifacoum poisoning and it also had blood on the back of the head consistent with fighting.

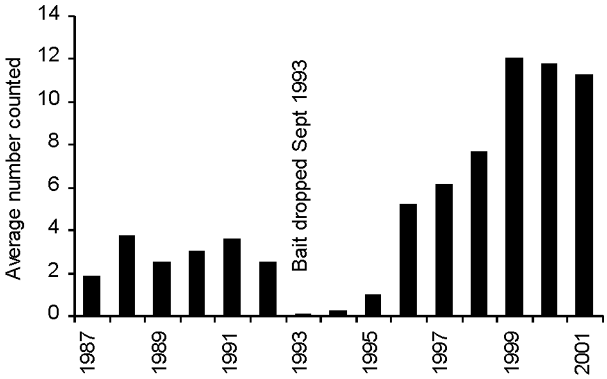

Fig. 2 Counts of pukeko on all transects in November of each year (OSNZ data).

The pukeko population declined by more than 90% in the weeks following the operation. Subsequent observations and counts by the Ornithological Society show that this population increased to higher than pre-poison drop numbers within two years of the operation (Fig. 2).

The robin population was estimated to be 40 birds at the time of the poison drop. About 11% of these died at the time of the poison drop and this mortality was attributed to the poison. This had no impact on population viability (Armstrong and Ewen 2000).

The colour-banded cohort of saddlebacks in Wattle Valley has been observed, and new birds have been banded since their introduction to the island in 1984. Natural mortality has been low and the population has continued to increase. Following the poison bait drop about 21% of this colour-banded population was lost (B. Walter pers. comm.). One bird was found dead and it contained brodifacoum (Table 1). The Ornithological Society counts also recorded a reduction in their counts through Wattle Valley but subsequent counts show that this has not been detrimental to the saddleback population in the medium term (Fig. 3).

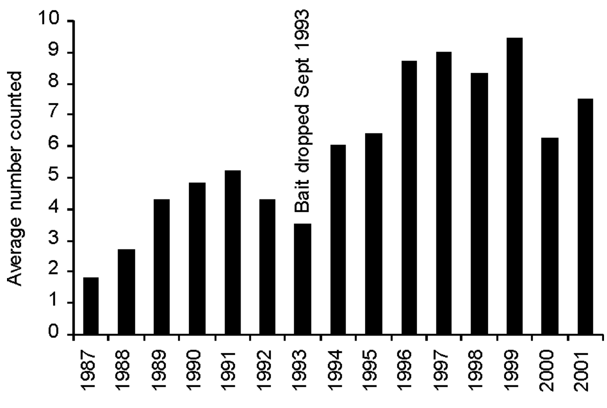

Fig. 3 Counts of saddlebacks on the Wattle Track transect in November of each year (OSNZ data).

The brown teal were released from their aviary on 11 November, six weeks after the bait drop. This species is not very conspicuous so their survival cannot be quantified. One pair, at least, raised young during the following summer of 1993/94.

The four takahe were released from their enclosures on 3 December, nine weeks after the bait drop. They all survived. Other takahe have been introduced to the island since the eradication operation and the takahe population continues to thrive.

Bellbirds (*Anthornis melanura*), tuis (*Prosthemadera novaeseelandiae*), parakeets (*Cyanoramphus novaezelandiae*), and saddlebacks all appear to have increased in number since the removal of rats from this island (Graham and Veitch 2002).

DISCUSSION

There was no sign of rodent presence in two years following completion of this operation. The operation has therefore been declared successful and there has also been no sign of rodents on the island since then. There is no evidence to suggest that the methods used have been detrimental to any non-target population in the longer term.

Before the operation began we knew it would not be possible to check on the continuity of bait spread, due to the nature of the vegetation on the island, but we did consider it possible to check the density of the bait spread using the methods described. The results of this work show that using the roads was not an effective way to check bait spread. The polythene sheet on the takahe pen was an effective way to check bait density but many replicates of this would be needed to ensure that there were adequate samples from areas where two flight lines did not overlap.

No samples of the fresh bait were checked for brodifacoum content. The bait found on 2 November, 35 days after the drop, contained brodifacoum at 22.1 ppm which was within the expected range. The bait in the decay plots was found to contain 30.6 ppm brodifacoum after 53 days of weathering. For this trial a portion of the bait used should have been stored for assay alongside the trial baits. As this was not done there is no way of knowing whether the 30.6 ppm was a high toxin load to start with, was caused by subsequent change in bait composition, or was an error in the assay process.

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Eradication of alien plants on Raoul Island, Kermadec Islands, New Zealand

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Abstract In order to protect the endemic ecosystem of Raoul Island (2943 ha), a programme to eradicate alien plants commenced in 1972. Almost 30 years on it is possible that seven species have been eradicated, none of which was widespread but some of which were difficult to control. There are another 22 species on the eradication list, most of which are barely present. Seven species represent the greatest threat at present and also are the most difficult to control. These are *Senna septemtrionalis*, *Caesalpinia decapetala*, *Anredera cordifolia*, *Psidium cattleianum*, *P. guajava*, *Olea europaea* subsp. *cuspidata*, and *Passiflora edulis*. Difficulties of the programme include the rugged terrain, resistance of some species to herbicide, cryptic species, and long-lived seedbank of some species. Each year an area equivalent to one quarter of Raoul Island is grid-searched twice; this is the area where alien plants are known to be present. The remainder of the island is checked during the recreational time of staff and volunteers and occasionally by air. The alien plant eradication programme has been successful to date but still has many years to run. Changes in abundance or distribution of some alien plant species are expected as a result of the planned rat eradication in 2002 and, in anticipation of the changes, a number of species have been eradicated. Maintaining search efficiency and staff morale at low alien plant densities and determining the end point of the programme will be a challenge.

Keywords Conservation; endemic species; rat eradication; grid searching.

INTRODUCTION

Raoul Island is the northernmost (29° 15' S, 177° 55' W) and largest island of the Kermadec Group which lies within the central Polynesian biogeographic region (Udvardy 1975). The bulk of the island was first formally protected as a Fauna and Flora Reserve, along with the rest of the Kermadec Group, in 1934 (Devine 1977). It is currently designated as a Nature Reserve and is surrounded by a marine reserve. Since 1987 the New Zealand Department of Conservation has been responsible for management of Raoul Island and maintains a small permanent staff there, who undertake alien plant control, weather and seismological recordings and general maintenance of facilities.

Raoul is an active volcano 2943 ha in extent and rising to 516 m above sea level. There is a central crater containing three small lakes (Fig. 1) which is encircled by the crater rim, rising from 40 to 516 m. The last eruption, in 1964, resulted in the formation of a new, small crater within this central crater. From the crater rim, ridges >300 m a.s.l. run south (Mahoe Ridge) and west (Hutchison Ridge). On

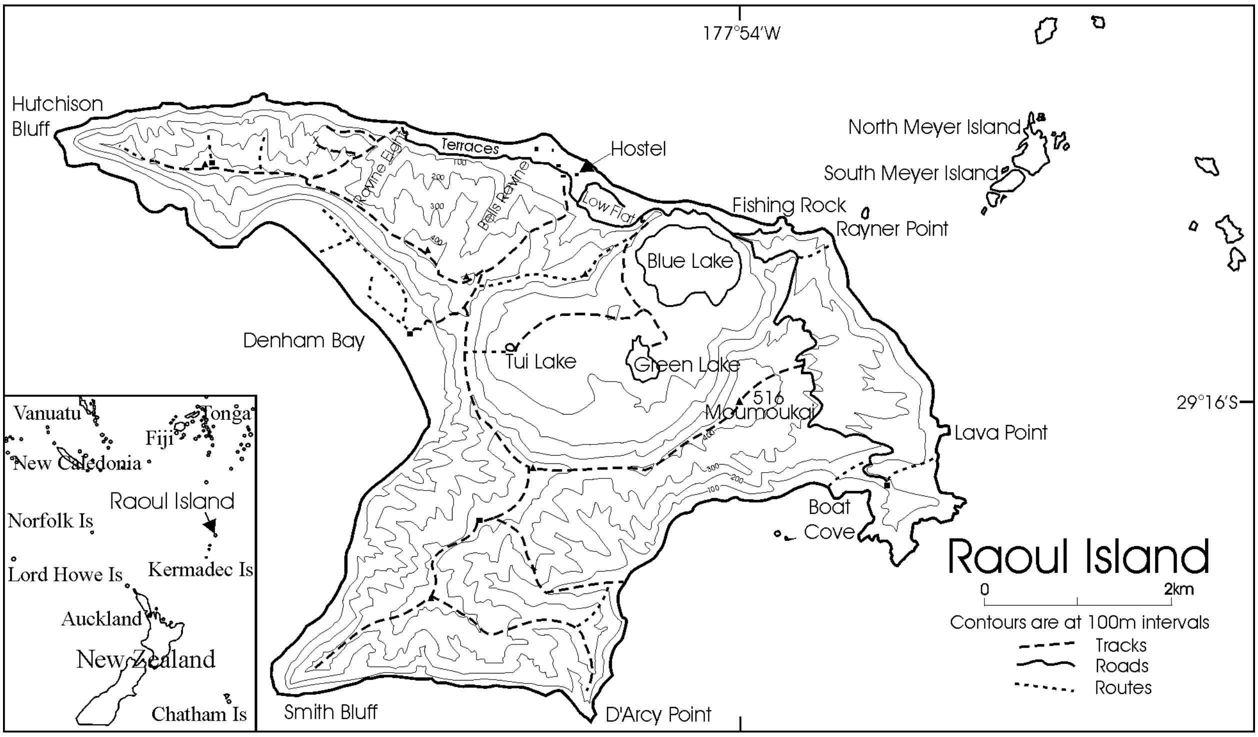

Fig. 1 Map showing features of Raoul Island mentioned in the text.

Pages 365-373 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

the north side of Hutchison Ridge deep ravines with no permanent running water reaching the sea. During periods of heavy rain, however, the ravines become torrents. Permanent seepages occur at just three points on the island and there is a swamp at Denham Bay. The terrain is steep and much of the coast is cliff-bound. The climate is warm temperate: average annual rainfall is 1538 mm, evenly spread throughout the year; mean annual temperature is 19°C; humidity is generally >80% (New Zealand Meteorological Service 1983). Tropical cyclones may occur during the summer and have a strong modifying effect on the forests of Raoul Island (Sykes 1977).

Forest dominated by *Metrosideros kermadecensis* is the main vegetation on Raoul Island. Above 300 m altitude is "wet forest" where the principal understorey species is *Ascarina lucida* var. *lanceolata* in association with *Rhopalostylis baueri* var. *cheesemanii*, *Homalanthus polyandrus*, and *Pseudopanax kermadecensis*. The wet forest lies within the cloud zone and collects moisture from the mist. At lower altitudes is the "dry forest" where the understorey is principally *Myrsine kermadecensis*, *Coprosma acutifolia* and *Macropiper excelsum* var. *psittacorum*. There is a narrow coastal zone where *Myoporum kermadecense*, *Cyperus ustulatus*, and *Isolepis nodosa* are common.

Catastrophic natural disturbances have occurred frequently during the history of Raoul Island. Volcanic eruptions have caused the total destruction of vegetation within parts of the crater and defoliation of plants elsewhere (Sykes 1977; Lloyd and Nathan 1981). Earthquakes are common and may cause slips and vegetation loss. Cyclones frequently topple trees. In some seasons two or three cyclones may strike whereas in other seasons there are none. Flash flooding also can destroy vegetation and cause massive slips on the steep slopes (Bell 1911).

Humans have visited and occupied Raoul Island for brief periods since c. 960 A.D. (Anderson 1980). Evidence to support periodic occupation by Polynesians since this date includes the discovery of many adzes, some prehistoric settlement sites which include obsidian flakes from New Zealand and the presence of *Rattus exulans* and Pacific plants such as *Aleurites moluccana* and *Cordyline fruticosa* (Sykes 1977; Johnson 1991).

The European discoverer of Raoul Island was Captain Sever on the *Lady Penrhyn* in 1788. During the early part of the 19th century Raoul Island was used as a source of water and wood by whalers exploiting the lucrative "French Rock Ground" for sperm and southern right whales (Sykes 1977; Johnson 1991). European settlement of the Island began in 1836 with James Reed and his family plus others. Goats (*Capra hircus*) and pigs (*Sus scrofa*) had already been introduced and established on Raoul by 1836 (Straubel 1954), presumably by whalers. Cats (*Felis catus*) were possibly introduced by whalers also (Gabites 1993). From 1836 until 1870 other family groups settled on the island, mainly to provide supplies to the whalers (Johnson 1991). From 1878–1914 Thomas Bell and his family were resident, and from 1889–1892, a New Zealand Government Settlement Scheme resulted in four families and eight individuals arriving on Raoul to establish gardens for the provision of fruit and produce to the New Zealand mainland (Johnson 1991).

From 1914–1935 Raoul Island was generally uninhabited apart from a brief five month period in 1926–1927 when Alf Bacon and two others arrived to grow crops (Venables 1937). The wreck of the *Columbia River* in 1921 near Boat Cove is assumed to be the source of the *Rattus norvegicus* which abound on Raoul Island (Merton 1968). In 1935 Bacon returned to the island and a further group, including Venables, settled on Raoul from 1936–1937 (Venables 1937). In 1937, a Government expedition led to the establishment of a meteorological station (Davison 1938; Sykes 1977) which still operates.

Currently Raoul Island is staffed by a team of five Department of Conservation rangers who are employed on one year, non-renewable contracts. A changeover period of 1-2 weeks in November each year allows knowledge and skills to be passed from one team to the next. Each year a team of 6–10 volunteers arrives to assist with the alien plant eradication programme for up to four months, usually during the winter.

Goats were eradicated by 1984 (Sykes and West 1996) after a prolonged Government-funded campaign that began in 1937 and intensified in 1972 (Parkes 1984). The goats were abundant and had a strong modifying effect on the vegetation of the island, including some of the alien plant species. Pigs, which were never abundant, were eliminated from Raoul Island in 1966 (Sykes 1977).

As a consequence of human occupation of Raoul Island, approximately 64% of the vascular plant flora comprises alien species. The total number of vascular plant species recorded for the Kermadec Islands is 317, of which 22 species are endemic (19% of the total native vascular plants). Many of the alien species have no impact on natural vegetation processes but since 1972 a number of species have been targeted for eradication (Devine 1977). Some of the alien plant species also have historic value, principally the tree species that were valued by the settlers as food sources. There is potential conflict between the requirement to preserve resources of historic significance and any impact that some of these species might have on natural ecosystems. In addition, the effects of *Rattus exulans* and *R. norvegicus* on alien plant species growth and regeneration must be considered as these species will be eradicated (along with cats) within the next few years. The history of the alien plant eradication programme is described in this paper and some of the technical difficulties are discussed.

METHODS

Alien plant categories

Since the beginning of the alien plant control operations on Raoul Island, all alien vascular plant species have been categorised according to the degree of threat they pose to the natural environment. In addition, some species have been protected because of their historic significance (e.g., a range of citrus species planted by the early European settlers and a group of *Araucaria heterophylla* planted by Thomas Bell). The first classification, in 1972, grouped the alien species into seven categories which included eradication, control, surveillance and protection (Devine 1977). Category A included those "species which so threaten (whether actually or potentially) the preservation of the natural state that their extermination is a desirable and feasible goal", and included six species (Table 1). The species in the other categories were not listed.

Revisions of the alien plant categories were undertaken in 1982, 1992 and 1996 (West 1996). Some species were substantially reclassified (e.g., *Hibiscus tiliaceus* and *Alocasia brisbanensis*), whereas others have always been targeted for eradication (e.g., *Caesalpinia decapetala* and *Psidium* species) (Table 1). The number of categories has been gradually reduced from seven to three, with the three categories defined in 1996 corresponding to eradication (A), effect on ecosystem minimal (B), and historically significant plants (C). In Category A, 17 species were recommended for eradication (West 1996).

Some reclassification is now warranted, based on improved knowledge of the behaviour of some of these species and because of the discovery of new alien plant species. For example, in 1998 *Selaginella kraussiana* was discovered on Raoul Island and was immediately targeted for eradication.

Table 1 Categorisation of target alien plant species on Raoul Island from 1972 to the present. Category definitions are given in the footnote.

Species	Year			
	1972	1982	1992	1996
Caesalpinia decapetala	A	A	A	A(i)
Psidium cattleianum	A	A	A	A(i)
Psidium guajava	A	A	A	A(i)
Olea europaea subsp. *cuspidata*	A	A	A	A(i)
Furcraea foetida	A	A	A	A(i)
Hibiscus tiliaceus	A	A	C	C
Senna septemtrionalis		A	A	A(i)
Passiflora edulis		A	A	A(i)
Anredera cordifolia		A	A	A(i)
Foeniculum vulgare		A	A	A(ii)
Alocasia brisbanensis		B	B	B
Stenotaphrum secundatum		B	B	B
Aleurites moluccana		C		C
Populus nigra		C	C	A(ii)
Araucaria heterophylla		C		A(i), C
Ricinus communis		C	C	A(i)
Gomphocarpus fruticosus		C	C	A(ii)
Phormium tenax		C		
Brachiaria mutica		C	C	A(i)
Cordyline fruticosa		D	D	C
Colocasia esculenta		D		
Prunus persica		D	D	C
Vicia sativa		E	C	B
Trifolium campestre		E	C	B
Senecio jacobaea		E	A	A(ii)
Cortaderia selloana			A	A(i)
Tropaeolum majus			C	B
Brugmansia suaveolens			C	C
Phyllostachys aurea				A(i)
Cirsium vulgare				B
Bryophyllum pinnatum				B
Vitis vinifera				C
Phoenix dactylifera				C

Location and marking of alien plant infestations

Raoul Island has been divided into 13 weeding blocks covering 24.6% of the island's area and including part of the Meyer Islands. The blocks delimit the area in which the target alien plants are known to be present and are located along the northern and eastern faces, the crater,

Footnotes to Table 1.

1982: Category A – weeds where threat is reversible and covered by a current programme for extermination. B – weeds where plant invasion is irreversible; no control provided for in current programme. C – adventives which are a potential threat and are included in the current programme for surveillance and/or limited control. D – persistent relics of cultivation either of historic significance, a landscape feature or providing edible fruit which may be protected. E – New or recent arrivals which can be exterminated by a short-term operation initiated under the programme before they become naturalised.

1992: Category A – species which so threaten (whether actually or potentially) the preservation of the natural state that their extermination is a desirable and feasible goal. B – species which so threaten the natural state that their extermination is desirable, but is not feasible at the present time. C – adventives resulting from accidental or deliberate introduction which are a potential threat and are included in the current programme for surveillance. D – persistent relics of cultivation either of historic significance, a landscape feature or providing edible fruit which might be protected.

1996: Category A(i) – species which are known to have the potential to significantly alter the structure and composition of the native vegetation of Raoul Island in the long term and which must be eradicated. A(ii) – species which are unlikely to have long-term significant impact on the structure and composition of the native vegetation of Raoul Island but which are of sufficiently low abundance to be eradicated. B – adventives resulting from accidental or deliberate introduction which have no historic significance and which pose a minimal or no threat to the forest ecosystem of Raoul Island. C – persistent relics of cultivation of historic significance or providing edible fruit which may be protected.

Denham Bay and at D'Arcy Point (Fig. 1). With the exception of D'Arcy Point (where an infestation of *Senna septemtrionalis* was found in 1998), the distribution of these blocks reflects the past and present areas of major human use. The blocks contain from one (D'Arcy Point) to 30 plots (Denham Bay), ranging in extent from 0.01 ha to 54 ha. The large plots are termed null plots and are the balance of the area within a block that is not part of an active plot. Active plots are typically <1–<10 ha in extent. Active plots are searched at least twice each year and null plots are searched at least once in two years.

Grid searching is done by teams of 2–5 people walking along parallel lines 6–10 m apart, depending on understorey density, and stopping frequently to search the ground, subcanopy and canopy for alien plant species. The second grid search of a plot is done at right angles to the first where terrain permits. When an infestation is discovered it is marked with flagging tape and an estimated location is marked on a map. Grid searching provides for the most unbiased coverage of ground which is difficult but not impossible to traverse on foot. If this method was not used, workers would take easier route options and would most likely miss target alien species.

Not all terrain is suitable for grid searching, and inspection of the cliffs behind Denham Bay is done routinely using binoculars or a telescope from the flats below and from vantage points on the ridge above, especially when *Caesalpinia decapetala* is in flower (June–November). The cliffs behind Denham Bay are accessed using fixed ropes and abseiling equipment. Elsewhere on the island, areas of bluffs are searched and weeded from ropes (e.g., some parts of the *Anredera cordifolia* site at Fishing Rock and areas below *S. septemtrionalis* infestations).

Periodically, aerial surveillance is undertaken by helicopter if there is an opportunity when the Royal New Zealand Navy is involved with re-supply. Some flowering trees of *S. septemtrionalis* have been discovered in this way and aerial surveillance can also be useful for locating mature vines of *Passiflora edulis* or trees of *Olea europaea* subsp. *cuspidata*.

Control methods

All of the species targeted for eradication were initially highly visible with one or more dense areas of infestation. For almost all species the initial knockdown phase required considerable time spent cutting mature individuals and painting stumps with herbicide or scattering herbicide granules around them. Blanket spraying of dense, accessible infestations was undertaken and in the case of *Caesalpinia decapetala*, helicopter application of herbicide was used. Burning of areas of *C. decapetala* vines and fern-covered clearings was also undertaken, to hasten the decline of the seeds of this legume from the seedbank.

Following the initial control of dense infestation areas, grid searching in and around the areas of known alien plant infestation was used to detect most species. Most time is spent searching and when an area of dense seedling regeneration is encountered, considerable time can be spent hand-removing seedlings and adolescent plants. When a mature plant is encountered, all fruits are removed for destruction by burning, the stem is cut, and herbicide is applied to the cut stem.

For species with a persistent seedbank such as the legumes *C. decapetala* and *Senna septemtrionalis*, the ground cover is often cleared to maximise the amount of light reaching the soil and to promote germination of seeds from the seedbank. The potential longevity of the seeds of these species is not known but is estimated to be decades, so any disturbances which expose seeds to the light should expedite eradication of these species so long as there are no fresh inputs of seed to the seedbank.

Some species can resprout vigorously from cut stumps despite application of herbicide. Examples are *Psidium guajava*, *Ficus macrophylla* and *Prunus persica*. Return visits to treated stumps, and repeated cutting and herbicide application, are required to kill these individuals. The change from Tordon® granules to a wet herbicide mix has resulted in less regrowth from some of these species.

Eradication of *Anredera cordifolia* follows the same general pattern as the other species in terms of effort in relation to different activities but the detail of the method is different, given the biology of this species. On Raoul Island, *A. cordifolia* flowers but does not set seed, so there is no need to search for seedlings away from the current known sites. However, the species produces abundant, herbicide-resistant tubers which can disperse by gravity or water. After many trials, a herbicide formula which effectively kills foliage and stems, but not tubers, has been found (Escort® = metsulfuron 50 g, Roundup® = glyphosate 2 *l*, Pulse® = penetrant plus marker dye in 100 *l* of water). This is used to kill the foliage and thereby make the tubers more visible for hand removal. It is also hoped that repeated application of this mixture will eventually kill all regrowth from tubers that cannot be reached because of inaccessible terrain. Tubers are removed from the vines and soil surface and dug up with forks and hand trowels, packed carefully into large plastic bags inside large backpacks (designated for this job only), carried up a steep face to the road and transported back to the base. There the tubers are loaded into a desiccator that uses the heat from the generator exhaust for rapid desiccation. As a final precaution the dried tubers are burned in a fire pit nearby.

Reporting on alien plant infestations

In the early years of the alien plant eradication programme, reporting was variable; generally not a lot of detail was given but some information on the magnitude of the work undertaken could be gained. In recent years reporting has been more prescribed and has moved from text files in WordPerfect to an Access database (implemented in 1997).

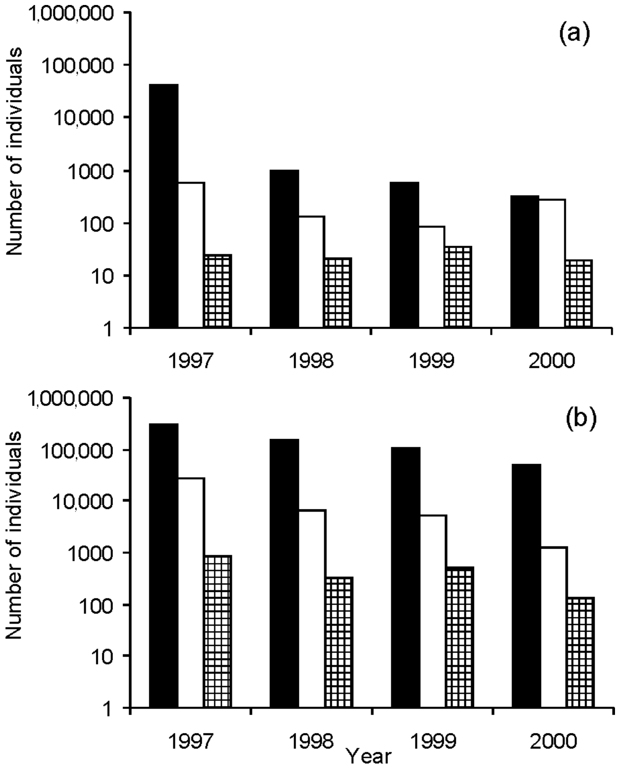

Fig. 2 Number of seedlings (black bars), adolescent (white bars), and mature plants (hatched bars) of (a) *Caesalpinia decapetala* and (b) *Senna septemtrionalis* removed from 1997–2000. Note the values on the y axis have been log transformed.

Key information recorded is the date, location and extent and age classes of an infestation, which species, action undertaken and the number of person-hours required to search and to treat each plot.

RESULTS

Each year the weeding blocks, occupying one quarter of the total area of Raoul Island, are grid-searched twice. In addition staff are always looking for alien plants when they are exploring the rest of the island during their time off (this is how the *S. septemtrionalis* infestation at D'Arcy Point was discovered). In the last six years, the staff have spent from 35–45% of their work time on weeding. In addition there is a huge boost to the eradication programme each year when 6–10 volunteers contribute labour for up to four months. The assistance of volunteers has approximately doubled the weeding effort in the past three years. The total weeding effort per year ranges from 7000 to 9000 person hours, including volunteer hours.

All alien plants targeted for eradication have been reduced considerably in number and extent. For example, *Caesalpinia decapetala* once occupied 22 ha of Denham Bay (Devine 1977) but in the last two years only 600–700 plants have been located and removed; the majority were seedlings (Fig. 2). Similar results have been obtained for *Senna septemtrionalis* which is the most widespread and still the most abundant target alien plant on the island. It occurs in 72% of the alien plant plots and it is the only target species on the Meyer Islands. Since 1997 the number of individuals located and killed has declined from >300,000 to just over 50,000 – a substantial reduction (Fig. 2). Although seedlings comprised the majority of the plants removed, many mature plants are still being found (e.g., 517 in 1999, 128 in 2000) (Fig. 2).

The database contains 29 species of which seven might by now have been eradicated (Table 2). *Furcraea foetida* occupied up to 0.5 ha in Denham Bay and *c.* 150 m^2 in the Dry Crater when eradication commenced in 1974. By 1982, only 11 plants were found in Denham Bay and in 1991, 12 plants were removed from the Dry Crater. In 1994, two terrestrial and one epiphytic plant were found in Denham Bay and several were epiphytic on *Metrosideros kermadecensis* in the Dry Crater (West 1996). In 1997, one adolescent plant was removed from Denham Bay. This species has not been seen since and, as it only ever reproduced vegetatively, it is possible that it has now been eradicated from the island.

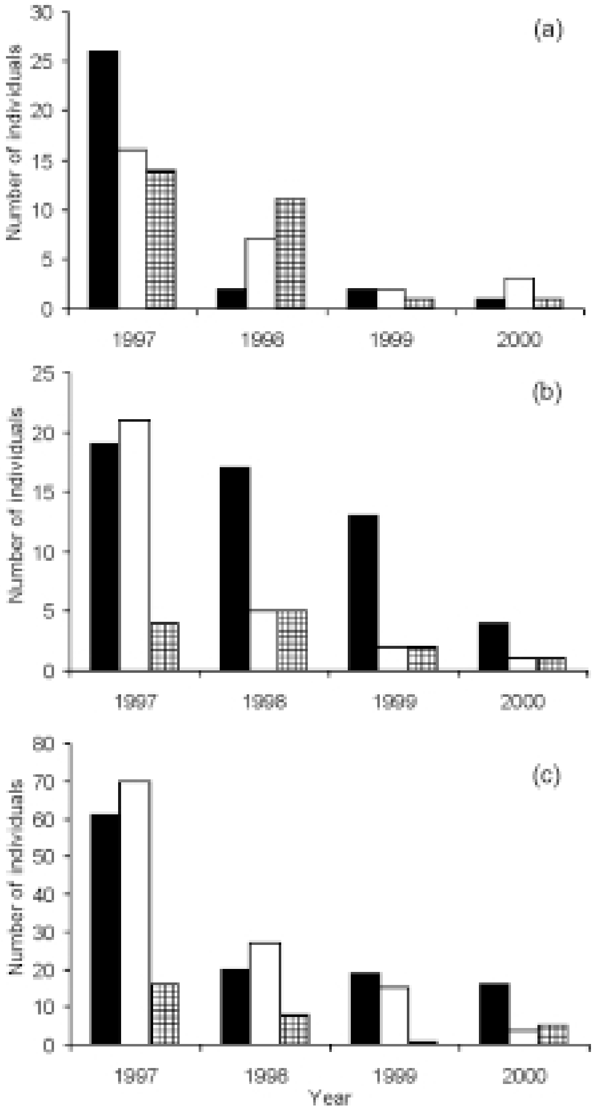

Fig. 3 Number of seedlings (black bars), adolescent (white bars) and mature plants (hatched bars) of (a) *Olea europaea* subsp. *cuspidata*, (b) *Psidium guajava* and (c) *P. cattleianum* removed from 1997–2000.

Table 2 Species listed in the alien plant database indicating when eradication commenced, whether they might have been eradicated, the total number of plants removed from 1997–2000 and the percentage of alien plant plots occupied by each species. * indicates that the measure used is kg.

Species	Common name	Eradication commenced	Eradicated?	Total number removed 1997–2000	% plots
Aleurites moluccana	candlenut	1993	no	827	2.1
Anredera cordifolia	Madeira vine	1995	no	*7410.6	2.1
Araucaria heterophylla	Norfolk pine	1974	no	959	1.4
Brachiaria mutica	Para grass	1996	no	97	0.7
Bryophyllum pinnatum	airplant	1998	no	4550	1.4
Caesalpinia decapetala	Mysore thorn	1974	no	44,877	18.2
Cortaderia selloana	pampas grass	1984	yes?	0	0
Ficus carica	fig	1996	no	43	2.8
Ficus macrophylla	Moreton Bay fig	1996	yes?	1	0.7
Foeniculum vulgare	fennel	1969	no	5	0.7
Furcraea foetida	Mauritius hemp	1974	yes?	1	1.4
Gomphocarpus fruticosus	swan plant	1979	no	9	2.1
Macadamia tetraphylla	macadamia	1996	yes?	17	1.4
Olea europea subsp. *cuspidata*	African olive	1973	no	86	19.6
Passiflora edulis	black passionfruit	1980	no	10,666	32.9
Phoenix dactylifera	date palm	1995	no	2	1.4
Phyllostachys aurea	bamboo	1996	no	1561	0.7
Populus nigra	poplar	1995	yes?	0	0.7
Prunus persica	peach	1994	no	4309	27.3
Psidium cattleianum	purple guava	1973	no	262	22.4
Psidium guajava	yellow guava	1972	no	94	11.9
Ricinus communis	castor oil plant	1990	no	208	7.7
Selaginella kraussiana	selaginella	1999	no	173	0.7
Senecio jacobaea	ragwort	1980	yes?	0	0
Senna septemtrionalis	Brazilian buttercup	1978	no	637,419	72
Tropaeolum majus	nasturtium	1999	no	2792	8.4
Vicia sativa	vetch	1996	no	1884	0.7
Vitex lucens	puriri	1997	yes?	3	0
Vitis vinifera	grape	1995	no	811	8.4

Cortaderia selloana was sown on Raoul Island to stabilise a retaining wall that was built to support the foxway winch shed at Fishing Rock. It was first recorded growing on that site in 1976 and all plants were removed when discovered in 1984 although, by then, one plant had flowered. If that was a hermaphrodite plant, some viable seed could have been set. Since 1984 at least 10 plants have been removed; one of these had flowered (Sykes and West 1996; West 1996). The last adolescent plant was removed in 1993 and none has been found since. Again, it is possible that this species has been eradicated from Raoul.

Some of the species which may now have been eradicated were relics of cultivation that were present in very low numbers but had the potential to spread once rats were eradicated. Examples include *Macadamia tetraphylla* which had begun to spread despite the heavy fruit predation by rats. In the last four years 17 individuals have been destroyed (Table 2), including four seedlings. Three trees of *Vitex lucens*, a species indigenous to northern New Zealand but not to Raoul Island, were also removed (Table 2). Experience from Tiritiri Matangi Island in the Hauraki Gulf had shown that *V. lucens* was unable to regenerate in the presence of *Rattus exulans* but seedlings were abundant after these rats were eradicated from that island (pers. obs.). The one large tree of *Ficus macrophylla* was cut and poisoned because *Pleistodontes froggattii*, the small wasp that pollinates this species, has arrived in mainland New Zealand (Gardner and Early 1996). If this wasp were to arrive on Raoul Island it might not be detected quickly enough to stop spread of *F. macrophylla* by seed.

Some alien plant species have persisted at very low levels for many years. Examples include *Foeniculum vulgare* and *Gomphocarpus fruticosus* (Table 2) for which the total numbers removed in the last four years are five and nine, respectively. Both species grow near the hostel buildings and have been part of the eradication programme since 1969 and 1979, respectively (West 1996). Other species have persisted at low levels, including *Psidium cattleianum*, *P. guajava* and *Olea europaea* subsp. *cuspidata* (Table 2; Fig. 3). Less than 100 individuals of *P. guajava* and *O. europaea* have been found and killed in the last four years whereas >250 *P. cattleianum* have been removed. All species are comparatively widespread (Table 2) but with greatest concentrations along the slopes near the hostel. All have grown in the crater and *P. guajava* has been removed from Denham Bay.

Not all of the alien plants removed can be recorded as numbers of individuals because many spread vegetatively. Reduction in area occupied is a more useful measure of effort and success but this measure has not yet been implemented for species such as *Bryophyllum pinnatum* and *Phyllostachys aurea*. A useful measure for the removal of *Anredera cordifolia* is the mass of tubers removed and killed. In the last four years *c*. 7.5 tonnes of tubers have been removed from Fishing Rock and Bells Ravine (Table 2), with 80% of that total achieved in the last two years.

DISCUSSION

Although the decline in abundance of target alien plants is obvious to regular visitors to Raoul Island (Sykes and West 1996; Sykes *et al*. 2000), and data collected in recent years show steady declines in the number of individuals removed each year, the programme will still have to continue for many years. The future term of the programme cannot be estimated easily because there are many factors to take into account. In a recent review, Myers *et al.* (2000) outlined six requirements for successful eradication programmes. It is useful to assess the Raoul Island alien plant eradication programme against these requirements as they address some of the acknowledged challenges of the eradication programme.

The first requirement is that resources must be sufficient to fund the programme to its conclusion. For the last five years, the programme has cost approximately NZ$250,000 per annum. The bulk of staff time is spent maintaining facilities – buildings, generators, foxway, road, tracks, etc. Much of the plant is old and breakdowns are frequent. Significant capital is required to scale down some of the facilities and upgrade them to more environmentally sound alternatives. If this was done, the ongoing annual costs of the programme could be reduced considerably thereby improving the likelihood of its continuation. To date, resources have been sufficient to fund the programme.

The second requirement is that the lines of authority must be clear and must allow an individual or agency to take all necessary actions. This requirement is clearly met by the Raoul Island alien plant eradication programme as just one agency, the Department of Conservation, is responsible for management of the island. In the past, part of the island was gazetted as a meteorological station and farm. The New Zealand Meteorological Service had total jurisdiction over that part of the island – an area of 111 ha that was the centre of origin of most of the target alien plants. Also, before the formation of the Department of Conservation three other agencies had involvement with the management of the island. The Department of Lands and Survey administered the Fauna and Flora Reserve and undertook alien plant control, the New Zealand Forest Service eradicated goats, and the New Zealand Wildlife Service monitored the indigenous bird species.

The third requirement is that the biology of the target organism must make it susceptible to control procedures. The susceptibility of each of the 29 species listed in the database (Table 2) varies considerably. Seven species stand out as the top priority species for control (recorded by Sykes *et al.* (2000) as the main woody alien plants) based partly on the difficulty of their control. Those species are:

- *Senna septemtrionalis* – rapid maturity, abundant seed production, long-lived seedbank, some dispersal of seeds other than by gravity;
- *Caesalpinia decapetala* – rapid maturity, abundant seed production, long-lived seedbank, lots of nasty thorns;
- *Anredera cordifolia* – abundant herbicide-resistant tubers, located on unstable cliffs, potential to be dispersed to new sites by sea;
- *Passiflora edulis* – rapid maturity, abundant seed production, bird-dispersed seeds;
- *Psidium cattleianum* – very cryptic seedlings and adolescents, abundant seed production, bird-dispersed seeds;
- *Psidium guajava* – extremely persistent regrowth from poisoned stems, bird-dispersed seeds;
- *Olea europaea* subsp. *cuspidata* – cryptic foliage, regrowth from poisoned stems, bird-dispersed seeds.

Knowledge of the biology of each species and tenacity in developing techniques to eradicate them are keys to successful management. For example, developing the desiccator to destroy *A. cordifolia* tubers was a breakthrough since the previous method of breaking the tubers down in drums using a composting accelerant was not fast enough – the vines could grow and produce tubers faster than they could be killed. A further breakthrough is required to eradicate this species on bluffs that are too unstable to be reached safely. Because of the resistance of tubers to herbicide, a programme to aerially spray these bluffs would be lengthy and costly with no guarantee of success.

For the legumes with a long-lived seedbank, removal of the undergrowth and disturbance of the soil can hasten depletion of the seedbank. Also, natural disturbances resulting in soil movement can expose deeply buried seeds. Mass movement of soil and flood debris can extend the range of these species. Consequently, the regular searching of null plots is very important.

Given that Raoul Island is predominantly forested, and that most of the target alien plants inhabit the forest, bird dispersal of seeds makes control difficult. This is because it leads to the extension of ranges in an unpredictable fashion. It has been suggested that *Senna septemtrionalis* was most likely dispersed to the Meyer Islands by birds (Sykes 1977) even though birds are not the primary dispersers of this leguminous tree. It is very important, therefore, for staff to be vigilant for alien plants when they are recreationally exploring parts of the island which are outside of the regularly searched alien plant blocks.

The fourth requirement, according to Myers *et al.* (2000) is that re-invasion must be prevented. This is definitely possible for the target species on Raoul Island. Most of the species were introduced to Raoul Island deliberately as food plants, or for fencing (*Caesalpinia decapetala*) or ornamental purposes (*Senna septemtrionalis*). Few of the target alien plants represent accidental introductions (e.g., *Selaginella kraussiana* and *Vicia sativa)*. Importation of new plants to Raoul Island is prohibited, consistent with management of the island as a Nature Reserve. The garden vegetables and herbs grown by the staff have been screened for their potential as alien plants or vectors for disease which could affect native plant species. Accidental transport of alien plants is greatly minimised by current quarantine measures. All visitors to the island are required to have clean footwear, clothing, and baggage. No soil or vegetation of any sort, other than fruit and vegetables for consumption, may be taken to the island.

The fifth requirement is that the pest be detectable at relatively low densities. Herein lies one of the major challenges of the Raoul Island alien plant eradication programme. For alien plants which have never spread beyond their initial focus of infestation, detection is not a problem as the area to be searched is small and the chance of detecting the alien plant is high (e.g., *Cortaderia selloana*, *Foeniculum vulgare,* and *Phyllostachys aurea)*. However, for more widespread species, it can be a problem, especially when these species are cryptic (e.g., *Psidium cattleianum* and *Caesalpinia decapetala)*. The detection of some species is increased when they flower (e.g., *C. decapetala* and *S. septemtrionalis*), but the challenge is to locate and remove flowering plants before they release a new crop of seeds.

One of the downsides to employing staff on one-year contracts is that they may not be sufficiently familiar with the target alien plant species to recognise them, especially when they are present in very low densities. In an effort to counteract this problem, one of each of the target plants is retained at the base as "pet" plants for training of staff. Exceptions are *Caesalpinia decapetala* where a "pet" plant is retained at Denham Bay and *Anredera cordifolia* which is still abundant at Fishing Rock. In addition there is an alien plant manual with colour photographs and clear descriptions of the plants, highlighting the key features for identification.

In terms of job satisfaction for staff, one of the hardest things is to grid search for prolonged periods without encountering any alien plants. Staff begin to doubt their own abilities to detect alien plants and can lose motivation. At this stage, alien plants can be missed – even mature, seeding specimens – and if this happens the whole programme can be set back considerably. Each team usually arranges their roster to provide some variety in the tasks undertaken as a safeguard against complacency or grid-searching burnout.

One of the difficulties experienced in this eradication programme has been the short life of the flagging tape in recent years. The eradication teams have noted in their reports that the newer supplies of flagging tape break down very quickly in sunlight. Also, in one report it was stated that a piece of recently-used flagging tape was found stuffed in a knothole, slashed by so many rat bites that the writing was barely legible. Use of GPS should overcome the difficulties associated with relocating most of the alien plant infestations. Since 1 May 2000, accurate GPS fixes have been available without the use of ground triangulation systems (a system that was unaffordable for Raoul Island) and many receivers will now work quite reliably beneath a forest canopy. In future, therefore, GPS will be used for relocation of alien plant infestations on Raoul Island.

Determining when a species is eradicated is difficult. Seven alien plant species may have been eradicated from Raoul Island. Of those listed in Table 2, we could probably remove the question mark from *Ficus macrophylla*, *Populus nigra*, and *Senecio jacobaea* because only one individual of each was ever detected as part of the eradication programme. Those plants are dead and no progeny have ever been observed. In the case of *S. jacobaea*, the one plant was pulled out before it had flowered (West 1996). We could probably remove the question mark from *Macadamia tetraphylla* and *Vitex lucens* also, since there were only a small number of those present (three trees of *V. lucens*) and they had a clumped distribution. No *M. tetraphylla* have been found in the last two years. It is just possible that *Cortaderia selloana* and *Furcraea foetida* have been eradicated too. The former has not been recorded since 1993 and the latter since 1997. The difficulty with being able to declare when an alien plant species has been eradicated is that seeds can persist in the soil and germinate many years after the last live plant has been seen. Also, unlike animals, plants tend not to leave a lot of sign of their presence.

The sixth requirement is that environmentally sensitive eradication might require the restoration or management of the community or ecosystem following the removal of a "keystone" target species. This is unlikely to apply to the Raoul Island alien plant eradication programme because the target alien plants form a small part of the total plant biomass on the island. The few native bird species do not rely on any of the target species but the rat species are probably benefitting the most from the alien plants. In terms of a rat eradication programme, this sixth requirement has driven the removal of some plant species which were not spreading (e.g., *Vitex lucens*). Species such as *Vitis vinifera* were vegetatively spreading and occupying large areas and control of these has commenced to ensure that no vines will flower and fruit once the rats have been eradicated, as it appears that rats eat all of the fruit produced by these vines.

Many of the species of historical value have begun to spread and will spread further and faster once rats are eradicated (e.g., *Citrus* spp., *Araucaria heterophylla*). Any conflict between preserving these species *in situ* for their historical value and removing them because of their threat to the endemic species and communities of the island will have to be resolved before the rats are eradicated, so that scarce conservation resources are not wasted. All of the *Vitis vinifera* cultivars and *Citrus* species located on the island and assumed to be of historical value are in cultivation on the mainland. Thus, if these species must be removed from the island, they can be restored into another more manageable setting to tell the story of the human history of Raoul Island.

CONCLUSION

The alien plant eradication programme on Raoul Island is, of necessity, a long-term project. Enormous effort has gone into the programme to date; all of the original target species are now much reduced in range, and some species have probably been eradicated. The improvements made to location, marking, and recording of alien plant infestations in the last decade have all benefited the alien plant eradication programme. Grid searching is an effective method of locating target plants in areas of known distribution and GPS will enable accurate relocation of known infestation sites. Wider surveillance both aerially and terrestrially is also an essential part of the eradication programme. A re-evaluation of priorities for management may be needed, especially with rat eradication planned for 2002–2003. The requirements for successful eradication of alien plants can be met on Raoul Island so long as resourcing is sufficient, and capable, motivated staff and volunteers are selected for the programme. The results of the alien plant eradication programme so far have been positive and any reduction of effort could result in lost ground. The programme must continue.

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Removing cats from islands in north-west Mexico

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Abstract Feral cats have been associated with extinctions of endemic island species throughout the world. Removing cats from islands is an effective way to protect biodiversity, but compared to other invasive alien mammals, cats are difficult to eradicate. Here we describe the techniques we used to eradicate cats from 15 islands in north-west Mexico between <1 and 43 km^2. These eradication techniques were developed and refined on small islands (<1 km^2) and then adopted successfully on larger islands (1– 43 km^2). Experienced hunters and trappers, and high quality hunting dogs were critical for successful cat eradication. The most effective technique was trapping and the most critical components of trapping were trap design and placement.

Keywords Gulf of California; Baja California; trapping; conservation; feral cats; eradication; introduced species.

La remoción de gatos de las islas del noroeste de México

Resumen Los gatos ferales han estado asociados con la erradicación de especies endémicas en islas en todo el mundo. La remoción de estos gatos de islas es una manera efectiva de proteger a la biodiversidad. Sin embargo, en comparación con otras especies invasoras de mamíferos los gatos son difíciles de erradicar. Describimos aquí las técnicas que utilizamos para remover a gatos de 15 islas del noroeste de México entre <1 y 43 km^2 de tamaño. Estas técnicas de remoción fueron desarrolladas y refinadas en islas pequeñas (< 1 km^2) y posteriormente se adaptaron exitosamente a islas de mayor tamaño (1– 43 km^2). Cazadores y tramperos experimentados así como perros de caza de alta calidad fueron esenciales para la erradicación exitosa de gatos. La técnica más efectiva fue el trampeo y los componentes más críticos del mismo son el diseño y ubicación de las trampas.

INTRODUCTION

Although less than 20% of the Earth's animal species are restricted to islands, 75% of all recorded animal extinctions since 1600 have been on islands (World Conservation Monitoring Centre 1992). Many of these extinctions were caused, at least in part, by predation, competition, and habitat alteration from invasive alien species (Atkinson 1989; Diamond 1989). Invasive alien species continue to cause extinctions on islands today (Mellink 1992; Smith *et al.* 1993; Alvarez-Castaneda and Cortes-Calva 1996; Moran 1996; Grant *et al*. 2000; Cowie 2001; Veitch 2001). Fortunately, introduced species can be eradicated, even from large islands. For example, nutria (*Myocastor coypus*) were eradicated from Great Britain (233,000 km^2; Gosling and Baker 1989), rats (*Rattus norvegicus*) were eradicated from Langara Island, Canada (31 km^2; Taylor *et al*. 2000), and exotic herbivores are being removed from increasingly-larger islands (Towns and Ballantine 1993; Keegan *et al*. 1994b; Simberloff 2001). Many of these projects benefited from the development of a host of new poisoning and hunting techniques that have dramatically improved eradication techniques for goats (*Capra hircus*) (Taylor and Katahira 1988; Keegan *et al*. 1994a; Parkes and Macdonald 2002), commensal rodents (*Rattus* spp. and *Mus musculus*) (Taylor and Thomas 1993; Taylor, *et al.* 2000; Thomas and Taylor 2002), rabbits (*Oryctolagus cuniculus*) (Chapuis *et al.* 2001), and pigs (*Sus scrofa*) (Schuyler et al. 2002).

In contrast to some of the above invasive alien species, cats remain very difficult to eradicate from islands (Veitch 1985, 2001). The largest island where cats have been successfully eradicated is Marion Island, South Africa (290 km^2), a project that took over 10 years (Bloomer and Bester 1992; Bester *et al*. 2000). The second largest island where cat eradication has been successful is Little Barrier Island, New Zealand (28.1 km^2); a project that took three years after previous failed attempts (Veitch 2001). Reasons for the inherent difficulty of successful feral cat eradications include the lack of effective baits that are attractive to cats (Morgan *et al.* 1990) or innovative hunting techniques comparable to the Judas goat technique (Taylor and Katahira 1988). Consequently, managers have had to resort to the persistent use of an array of methods (Veitch 1985). The difficulty of feral cat eradication poses a significant problem to the conservation of biodiversity, since cats are widely distributed on islands and are associated with many extinctions and extirpations (King 1985; Atkinson 1989; Diamond 1989). Thus, more information is needed on the distribution, ecology, and behaviour of feral cats on islands; successful cat eradications from is-

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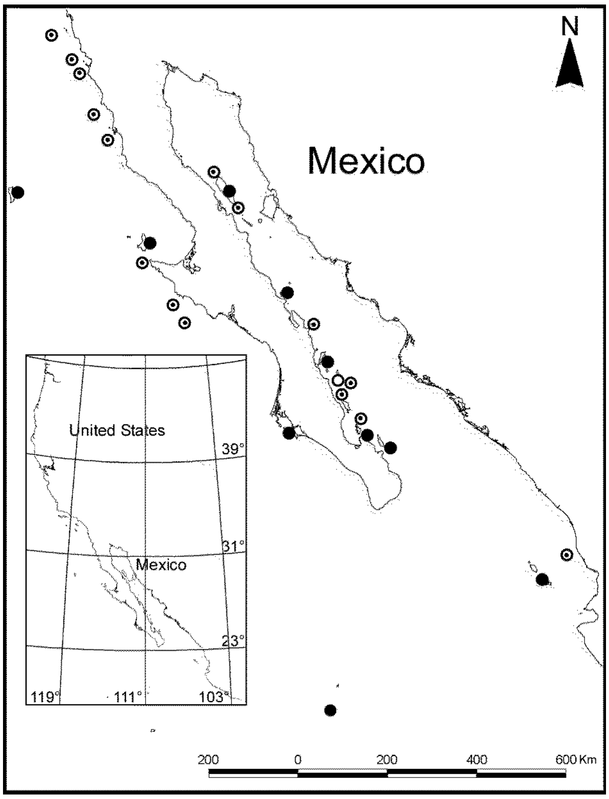

Figure 1 The distribution of feral cats on islands in north-west Mexico (circles). Cats have been eradicated from 15 islands since 1994 (circles w/ black dots), an additional island (Santa Catalina) is near completion (white circle). Cats remain on 10 islands (black circles).

lands; and the hunting, trapping, and poisoning techniques used.

Off the Pacific and Gulf of California coasts of north-west Mexico there are over 250 islands (Fig. 1). These islands are of both continental and oceanic origins and have numerous endemic reptiles, terrestrial birds, mammals, seabirds, and plants (Case and Cody 1983; Everett and Anderson 1991; Junak and Philbrick 1994a, 1994b; Alvarez-Castaneda and Patton 1999; Grismer 1999a; Donlan *et al*. 2000; Junak and Philbrick 2000). Cats have been implicated in several bird and small mammal extinctions and numerous seabird extirpations in the region (Jehl and Parkes 1982; Jehl and Everett 1985; Brattstrom 1990; Mellink 1992; Smith *et al.* 1993; Alvarez-Castaneda and Cortes-Calva 1996; Martinez-Gomez and Curry 1996; McChesney and Tershy 1998). Diet studies also indicate that cats may be impacting endemic reptile populations on some islands, although no reptile extinctions have been recorded.

In 1994, feral cats were present on 26 islands in north-west Mexico. In this paper, we describe the trapping and hunting techniques used to eradicate cats from 15 of those islands. These eradications were not conducted to test eradication methods and data were not collected on the efficacy of various techniques. Consequently, we are unable to present a quantitative evaluation of our techniques. Nevertheless, we feel that a detailed description of the methods used will be useful to those planning and conducting cat eradications.

The islands from which we eradicated cats are arid or semi-arid in climate, with precipitation ranging from <20-255 mm per year (with the exception of the subtropical island Isabela (600 mm rain/year); Hastings 1964; Hastings and Humphrey 1969). Vegetation communities on the islands of north-west Mexico include Mediterranean coastal chaparral, Sonoran desert, and dry subtropical forest (Shreve 1936; Levin and Moran 1989; Moran 1996; Esler *et al*. 1998).

METHODS

To summarise the distribution of feral cats on the islands of north-west Mexico, we relied on published and unpublished literature as well as personal communications from researchers and island residents, and our own field notes. We recorded the presence or absence of cats on each island. These data were then compiled in a conservation database accessible to the public (http://www.islandconservation.org; Donlan *et al.* 2000; Tershy *et al.* 2002).

To remove cats from islands we adapted hunting and trapping techniques used by the most successful commercial bobcat (*Lynx rufus*) trappers in the south-western United States of America. These techniques have traditionally been closely-guarded secrets and have not been subjected to scientific testing. They involve hunting, the use of dogs, and trapping. The techniques described here compliment those of Veitch (1985, 2001).

We did not conduct specific research on cat behaviour or diet prior to eradication efforts. Nor did we attempt to estimate cat population sizes. Rather, after thoroughly surveying the island for cat sign and trails, we began eradication efforts. Eradication efforts then continued until there was no evidence of new cat sign for several weeks to months depending on the size and complexity of the island. After each island was thought to be free of cats due to the absence of sign, at least two subsequent visits were made at three to eight month intervals to check for new cat sign. We considered cats to have been successfully eradicated from an island after it had been re-visited at least twice with no cat sign detected.

Hunting

We attempted to train biologists and inexperienced hunters to hunt cats, but had only limited success. Ultimately, the only effective hunters we employed had many years of experience hunting both at night and day. As night hunting is illegal in many areas of Mexico, former poachers are often the only individuals with these skills. We felt that in the process of learning to hunt effectively, inexperienced hunters ultimately trained cats to avoid hunters by missing or only wounding the cats. If so, this could make it more difficult to eradicate the last cats.

Hunting at night with .22 and .222 calibre rifles was more effective than hunting during the day. Hunters walked quietly using an adjustable headlamp set on low power to locate cats by their eye shine. Once a cat was located, the hunter often increased the power on the light to aid in aiming. We felt that high candlepower spot lights frightened cats from farther away than we were able to effectively shoot them (within about 100 m). Occasionally, in order to attract the cats' eyes towards the light, hunters made calls that mimicked cat prey. Hunting during the day was most effective with the aid of trained dogs. Hunting with shotguns was not as effective as hunting with rifles because shotguns could only be used within about 40 m from a cat, a distance often difficult to achieve.

Dogs

Acting on interviews with two successful mountain lion hunters, well-bred experienced dogs facilitated hunting and trapping. Out of the 12 dogs we field tested, we kept six; all of which were dogs bred for hunting and are valued at USD1000 – USD2000 each. Jack Russell Terriers were used exclusively because they were motivated to hunt cats and their small size facilitated both transportation between islands and maintenance of the dogs in the field. During the day, trained dogs tracked down and flushed out cats, or simply located areas where cats were present, which greatly aided in trap placement. On larger islands we believe that hounds trained for mountain lion (*Felis concolor*) hunting in the western United States could be especially useful. However, transport and field maintenance of these larger dogs will prove more difficult.

We attempted to train a number of dogs that were bred for show or as pets, but had limited success. We found that dogs from elite hunting lines were well worth the extra monetary investment because they were much easier to train, and performed more effectively than non-hunting dogs. Because there is no legal bobcat or mountain lion hunting culture in Mexico, we imported proven dogs from the United States.

Dogs are much easier to train when the density of cats is relatively high (i.e., at the beginning of an eradication campaign), rather than toward the end of a campaign when the density of cats is low. Also, one or two experienced dogs greatly facilitated the training of new dogs. Consequently, we prioritised buying and training top quality dogs early in our eradication programme.

Trapping

Well-located traps were much more effective than hunting, especially for removing the last cats. Traps work 24 hours per day, seven days a week, and could be checked from a distance with binoculars. We used Victor #1½ padded leg-hold traps. Larger traps can injure cats and paradoxically, make them more likely to escape; cats could possibly pull free from smaller traps. We prepared new traps by first cleaning them of oil and grease with hot water or a steam cleaner. Next we dipped them first in a commercial trap dip solution to slow corrosion and take away the shine and second in wax to further protect the traps and lubricate the moving parts. Finally, we carefully adjusted the pan tension and height of the pan on each trap to ensure that it had the correct sensitivity for young and mature cats.

We used two types of sets. These were 'cubby sets', where the trap blocked the single entrance to a cave or hole, with the bait or scent placed behind the trap, and 'walk through sets', where the trap was along a trail and the bait or scent was placed above or slightly to the side of the trap. These sets are conceptually similar to Veitch's (2001) "baited set" and "walk through set".

All sets had the same basic structure. The trap was placed so that the jaws opened parallel to the cat's direction of travel. To ensure the cat stepped on the trap, the path was narrowed by placing rocks, or other obstructions, as close as possible on either side of the trap without touching the trap jaws. This insured that the cat could not step on either side of the trap. The rock on the dog, or trigger, side of the trap formed a perpendicular wall about 14 cm tall. A rock on the pan side of the trap was approximately the same height, but slanted slightly away from the trap. This arrangement encouraged the cat to step on the pan side of the trap with its front foot. To further encourage this behaviour, a small amount of bait or scent was placed on the rock on the trigger side of the trap. More importantly, an obstruction of twigs or small rocks about 5 cm high (slightly lower on the pan side) and 4-6 cm wide was placed on the path in front and behind the trap. Cats avoided stepping on this guide and stepped over it directly onto the trap. The exact width of the guides was scaled to the stride of the cats as determined by tracks.

Overall, the funnels, rocks on either side of the traps, and guides formed a series of subtle obstacles that made it easier and more likely for the cat to step on the trap pan than anywhere else. Cats could easily jump over any of these obstacles, but, when constructed correctly, cats tended to walk through them and step on the trap pan. Because cats that spring a trap without being captured may become trap shy, our goal was to capture more than 90% of the cats that travelled through these trap sets.

Rather than burying the trap, we often simply placed a pan cover over the pan and left the trap jaws exposed. The pan cover increased the sensitive area of the pan, protected the trap from being jammed by blown debris, and provided a more natural surface for the cats to step on, without the need to cover the trap. We made a pan cover for each trap out of plastic or nylon window screen mesh. The mesh was cut into a 12 cm square with a 4 cm slit cut from the middle of one edge toward the centre of the square. A contact adhesive was then sprayed on the mesh and gravel or coarse soil (ideally of the same type found on the island) was poured over the adhesive. When setting the trap, this pan cover was placed over the pan, but under the open trap jaws.

Trap location was the most important element of success. We felt that one properly-placed trap was worth more than 10 poorly-located traps; subsequently, we spent much time scouting out keen trap locations on the island and designing trap sets. On some small islands, we caught the majority of cats in one location. Even on our largest project island (43 km^2), we had less than 50 traps deployed at any one time. We never attempted to trap on a grid system.

Traps were placed where cats were very likely to travel. To do this, we used the location of tracks and scats to guide trap placement. We especially trapped around "latrine areas" where one or more cats frequently defecated. We also tended to place traps along edges and natural restrictions where several trails came together (e.g., passes, trails through thick brush or high grass). Frequently, we narrowed natural restrictions with brush, rocks, and occasionally even 1m high plastic meshed fencing. Cats could easily jump over all these supplemental restrictions, but as long as they deflected the cats less than about 50 degrees from their direction of travel, cats tended to move along them and were funnelled into a trap. These funnels were designed to subtly guide cats into traps.

Scent, or occasionally bait, were used only to attract a cat from a few metres away, or to slow it down as it passed by the trap. Scent, made out of a mixture of cat faeces and urine with some glycerin added as a preservative, was used more often than bait because it remained attractive to cats longer and did not attract non-target species. Scent made from cats that do not live on the island (i.e., strangers) may be more attractive than scent made from cats living on the same island.

When fish or other bait was used, it was placed under a rock or bush to avoid attracting scavenging birds and direct sunlight. Bait had to be replaced every one to two days. Old bait was collected and disposed away from the set since cats can be repelled by spoiled bait (Veitch 2001). Small amounts of scent or bait were used because when large amounts were used, cats often rolled in the scent or bait. A cat rolling on a trap often results in the trap closing without catching the cat.

Our most experienced trapper (BW) usually selected ideal locations for traps and designed a series of trap sets in those locations. Once these tasks was completed, less experienced trappers were able to check and re-set traps while the more experienced trappers established new trap locations, or started work on another island. However, experienced trappers needed to periodically return to the island to scout new trap locations, design new sets in the same locations and check that traps were being properly re-set. This was especially true near the end of a project because it was often necessary to modify trap sets, baits, and scents in order to successfully trap the last remaining cats.

To increase the efficiency of checking the traps, a flag or stick was set loosely in the ground and the trap chain was wrapped around it. When a cat was captured and tugged on the chain, the flag or stick was knocked down. This system enabled us to check multiple traps from the boat or from distance with binoculars. In some hard-to-observe locations, we taped a wildlife radio transmitter with a magnetic on-off switch and a small magnet to the trap chain. We then doubled the chain, so that the magnet turned off the radio transmitter. When a cat tugged on the chain, the radio began transmitting, enabling us to check traps without directly observing them.

Toxins

Poisoning with 1080 can be used effectively to eradicate cats (Veitch 2001). We did not use toxins because we felt it was more difficult to attract a cat to toxic bait, than to step into a well-set leg-hold trap. However, on future operations where we are limited by the number of skilled trappers, incorporating toxins into our techniques may prove effective.

RESULTS

Feral cats were found on 26 islands in 1994 (Fig. 1). Island Conservation and Ecology Group, the Instituto de Ecología at the Universidad Nacional Autónoma de México, Centro de Investigaciones Biológicas del Noroeste, and the National and regional offices of Areas Naturales Protegidas collaborated with local people and community organisations to eradicate introduced cats from 16 islands (Fig. 1; Table 1). The operation on one of these islands, Santa Catalina, is still in progress.

DISCUSSION

Feral cats cause extinctions on islands (King 1985; Mellink 1992; Smith *et al.* 1993; Towns *et al.* 1997; Dowding and Murphy 2001). The most effective way to permanently protect island species threatened by cat predation is eradication, and subsequently prevention of re-introduction. Yet removing cats from islands is difficult (Bester *et al.* 2000; Veitch 2001). The methods we have developed to successfully eradicate feral cats from 15 islands in north-west Mexico are not infallible, but when applied correctly, can greatly facilitate conservation.

The four essential lessons that we learned regarding cat capture techniques are:

- use the most experienced hunters, trappers, and hunting dogs available
- focus on trapping in order to get the last cats
- study cat movements and behaviour in order to select ideal trap locations, and
- a few well-constructed sets in key locations are worth hundreds of poorly-located traps

Table 1 Islands from which feral cats have been removed and number of native taxa protected.

Islands (North to South)	Area (km^2)[1]	Breeding Seabirds	Endemic species and subspecies		
			Reptiles	Landbirds	Mammals
Pacific					
Coronado Norte	< 1	11 (3[5])	2	2	1
Todos Santos Norte	< 1	5 (1[5])			2 (1[5])
Todos Santos Sur	1	6 (1[5])	2	1[5]	2 (1[5])
San Martin	3.2	6 (3[5])	3		2 (1[5])
San Geronimo	< 1	5			1
Natividad	7.2	6 (1[5])			1
San Roque	< 1	6 (1[5])			1 (1[5])
Asuncion	< 1	7 (4[5])			
Gulf of California					
Mejia	3.0	3	2		2 (2[5])
Estanque	< 1	1	1		
Coronados	8.5	1	1		3 (2[5])
Monserrate	19.4	2	2		2 (2[5])
Catalina (incomplete)	43.1	2	8		1
San Francisco	2.6	1	2		2
Partida South	20.0	0	3		1
Isabela[4]	1.0	10			
TOTAL		72 (20) 6[2]	26 (20)[3]	3 (3)[3]	21 (18)[3]

[1] Areas are estimates based on literature.
[2] 72 seabird populations (20 seabird species and subspecies), 6 endemic to north-west Mexico.
[3] Number of endemic populations (number of endemic species and subspecies), some taxa occur on more than one islands.
[4] Island Conservation and Ecology Group assisted Cristina Rodríguez of Instituto de Ecologia, Universidad Nacional Autónoma de México; Norway Rats still present.
[5] Possible extinctions (extirpations for seabirds); e.g., 3 (2[5]) = 3 endemics, 2 of which may be extinct.

Using these techniques we have successfully removed cats from islands up to 20 km^2, and have removed most of the cats from a 43.1 km^2 island. We will soon be attempting to apply these same techniques to larger islands up to 250 km^2. On these islands research on cat home ranges, habitat use, and movement patterns will greatly facilitate trap spacing and placement. Furthermore, we may have to incorporate additional techniques such as disease and toxins to get an initial decrease in cat numbers as suggested by Veitch (1985) (Courchamp and Cornell 2000).

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The evolution and execution of a plan for invasive weed eradication and control, Rangitoto Island, Hauraki Gulf, New Zealand

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Abstract A plan for management of invasive weeds on Rangitoto Island, in the Hauraki Gulf, Auckland, New Zealand, was developed during five years of weed control on the island. Rangitoto is a shallow marine basaltic shield volcano, with gently sloping fragmented lava flanks topped by a central scoria cone. Invasive weed control aims to protect the native plant communities and the unique plant successional processes from bare lava to forest. There are 72 species of invasive weeds destined for control or eradication, many of which are not managed as weeds elsewhere in New Zealand. A draft plan was devised in 1995 that considered distribution, impact on the native vegetation, and efficiency of propagule dispersal in setting priorities for control. The result was a strategy with a top priority of controlling around 20 species that had the potential to drastically alter the natural vegetation, but still had very limited populations. The second stage of the strategy was to control the remaining 50 species on a geographical basis, proceeding from the least-infested areas to the most densely infested, generally dictated by the distribution of the widely-distributed alien invasive *Rhamnus alaternus* (Rhamnaceae). The weed management plan has evolved over five years with improving control techniques, new herbicides, new weed finds, and better mapping and relocation systems.

Keywords Rangitoto Island; invasive weed control.

INTRODUCTION

Rangitoto is a 2331 hectare volcanic island situated within the Hauraki Gulf, near the city of Auckland, New Zealand (Fig. 1). The island is administered by the Department of Conservation as a Scenic Reserve, and is a popular day trip destination.

As part of managing Rangitoto, the Department aims to minimise threats to the island's natural plant communities. The central cinder cone and broken lava flanks of Rangitoto support a mosaic of *Metrosideros*-dominated forest and scrub that are scientifically important as a living example of vegetation colonisation and succession from fresh lava to native forest. Following the recent eradication of introduced mammalian browsers (declared complete in 2000), the main threat to the island's vegetation communities is posed by invasive weeds.

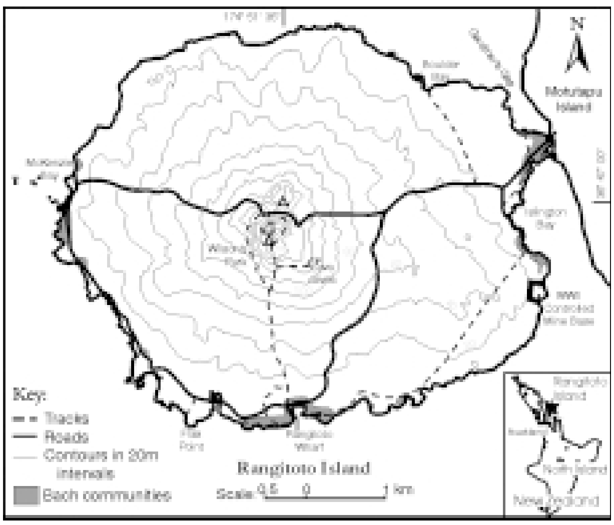

Fig. 1 Location and features of Rangitoto Island.

In 1995 funding became available for a weed management programme on Rangitoto. The large number of weed species meant that a detailed plan was needed to prioritise control action. This paper describes the process of developing a plan for managing the invasive weeds of Rangitoto. The plan evolved during five years of invasive weed control, going through three working drafts and becoming more comprehensive as our knowledge of the invasive weed problem on Rangitoto grew.

Natural History

Geology

The youngest and largest volcanic cone of the Auckland volcanic field, Rangitoto is described geologically as a shallow submarine Icelandic type shield volcano which formed during a single short eruptive event 600 to 700 years ago (MacDonald 1972; Julian 1992). Rangitoto is joined by a short causeway and bridge to the island of Motutapu, a much older landform which today is a pastoral farm with small pockets of forest.

Native vegetation

The forest structure and species associations found in the native vegetation on Rangitoto identifies more closely with the forests of the volcanic Big Island of Hawaii than with any other forest type in New Zealand. The canopy of the Rangitoto forests, like their Hawaiian counterparts, are dominated by trees of the genus *Metrosideros* (Myrtaceae).

Pages 381-388 In Veitch, C. R. and Clout, M. N. (eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Metrosideros excelsa (pohutukawa) and *M. robusta* (Northern rata) form a hybrid swarm which is thought to be progressively backcrossing to *M. excelsa* (Julian 1992). The forest ground layers on Rangitoto support many species which in northern New Zealand mainland forests grow as epiphytes, for example *Collospermum hastatum* (Liliaceae), *Griselinia lucida* (Cornaceae), and *Astelia banksii* (Liliaceae).

During the 600 years since Rangitoto erupted, native vegetation has colonised about 80% of the raw lava flows. The vegetation patterns of Rangitoto were described by Julian (1992) as being strictly governed by the underlying lava flow type. Briefly, the flows composed of large slabs or rafts of rock (close to the Hawaiian pohoehoe flows in nature) were colonised more quickly and today support a mosaic of continuous forest types, including the tallest and most well developed forest types. The fragmented (aa) flows support low scrub or vegetation islands dominated by *Metrosideros*. Some of these aa flows are still only 60% covered by vegetation, leaving large expanses of empty broken lava. If the natural colonisation process is allowed to proceed then it could be hundreds of years before the largest open aa lava areas become clothed in native forest.

Human history

Possums and wallabies

Possums (*Trichosurus vulpecula*) and wallabies (*Petrogale pencillata pencillata*) were introduced to Motutapu Island from Australia in the late 1800s. From Motutapu they quickly invaded Rangitoto. The combination of possums browsing in the canopy and wallabies browsing the lower forest tiers caused significant damage to the vegetation. Possums were consuming mainly *Metrosideros* foliage, in some areas defoliating the canopy so much that the ground tier plants beneath were dying. Both possums and wallabies were foraging widely on the ground, eating ferns, depleting shrubs and destroying seedlings (Julian 1992).

In 1990 possum and wallaby eradication commenced using aerial poisoning. This gave the vegetation immediate relief, as it resulted in a 90% reduction in possum and wallaby numbers. The eradication of possums and wallabies from Rangitoto and Motutapu Islands was declared complete in 2000 (Mowbray 2002).

While possums and wallabies were destroying the native vegetation of Rangitoto they were also keeping weeds in check. A possum and wallaby exclosure experiment established before possum and wallaby eradication began showed increases in the pampas grass (*Cortaderia selloana*), and prickly hakea (*Hakea sericea*) (Julian 1992).

Introduction of invasive weeds

There were several campaigns in the late 19th and early 20th centuries to "beautify" the harsh volcanic landscape of Rangitoto. Several planting days were held in the 1890s with members of the public encouraged to catch a ferry to Rangitoto, bringing plenty of their favourite plants out with them. A large range of garden plants were introduced by the three small bach (holiday home) communities which built up from 1911 to 1937 (Fig. 1). The most enterprising attempt at beautifying Rangitoto was by two Englishmen, Leary and Wilson, who planned a botanical park on the island, near the base of the summit cone. They went to considerable effort to bring cacti, figs, pines and paw-paws (Woolnough 1984). It is likely that Leary and Wilson also introduced *Erica lusitanica* (Ericaceae), and *Hypericum androsaemum* (Hypericaceae) which went on to establish in the summit cone area.

Approximately 60% of the plant species we now consider invasive on Rangitoto were probably introduced to the island as garden plants. The garden plants that have naturalised are typically succulent (e.g. *Crassula, Sedum, Aloe, Bryophyllum* spp.) or possess drought resistant features such as bulbs, corms or rhizomes (e.g. *Watsonia, Gladiolus, Iris, Nephrolepis, Asparagus* spp.).

The remaining 40% of invasive weed species have seeds distributed by birds, water, or light winds and could easily have made their way from mainland Auckland. Auckland has a large naturalised flora by New Zealand standards (615 species) (Auckland Regional Council 1998). The proximity of mainland Auckland (3 km) presents an ongoing weed invasion concern.

There are now at least 232 naturalised exotic vascular plant species on Rangitoto, compared with 286 native vascular species (Gardner 1997). Not all of the naturalised exotic species have become invasive, but many that have become invasive are relatively innocuous elsewhere, and this gives Rangitoto a weed flora that is unusual in New Zealand.

DEVELOPING A WEED MANAGEMENT PLAN FOR RANGITOTO

In 1995 funding became available for a weed control and eradication programme on Rangitoto. As a result of distribution surveys and consideration of biological characteristics as described below, we identified 72 species as having enough impact on the native vegetation to warrant management. These were the species which could significantly and adversely affect the long-term survival of native species, the integrity or sustainability of natural communities, or genetic variation within indigenous species (Owen 1998). With such a large number of weeds and limited resources it has been important to prioritise control actions and plan strategically.

Prior to this weed control programme, there had been no formal attempts at controlling weeds on Rangitoto, apart from several control operations concentrating on removal of wild pines (Segedin 1985).

Weed distribution surveys

Field surveys, a literature search, and advice from local botanists elucidated the naturalised species present, their distribution, impact or potential impacts, modes of spread, and controllability.

Field surveys were conducted over key sites to indicate invasive species distributions. Exotic naturalised species were surveyed and mapped:

- around the entire coastline, as this is the first point of land for birds bringing weed species from other sites and for wind-blown or sea-borne propagules to take hold;
- at sites of human occupation (i.e. quarries, baches and old bach sites and WWII military installations) for deliberate introductions and garden escapes;
- around roads and tracks, as bird highways, and because of the potential for people to carry weed seeds on clothing and vehicles;
- on the summit cone, having the most soil-like substrate and therefore supporting the greatest range of species;
- along three north-south transects through the island, following hunters' trap-lines to get an idea of what was in the interior.

The adjoining south-western third of Motutapu Island and the 13 residence gardens on Motutapu were searched for species which had the potential to spread onto Rangitoto.

Prioritising invasive weeds for management

Early in the development of the weed management plan, the invasive plant species were grouped into seven priority classes according to:

- the impact or potential impact they have on the native vegetation;
- how quickly they are able to spread; and
- their distribution across the island.

Degree of impact

The degree to which an invasive species impacts on native vegetation processes was accorded the most influence in setting priorities for control. We considered invasive species to have a high impact on native vegetation processes if they were:

- able to colonise the remaining bare lava, such as *Ulex europaeus* (Papilionaceae); or
- tree-sized when adult, often drought resistant or epiphytic species such as *Ficus rubiginosa* (Moraceae), *Rhamnus alaternus* (Rhamnaceae); or
- vines, able to climb and smother other vegetation, such as *Dipogon lignosus* (Papilionaceae); or
- able to form a dense ground cover that prevents the regeneration of native species, such as *Crassula multicava* (Crassulaceae).

Distribution and rate of spread

Most of the weeds on Rangitoto have limited distribution. Those species that were of limited distribution but had the potential to spread quickly (i.e. were in the colonisation 'lag' phase) were accorded a higher priority for control than those which we expected to spread slowly. There are some species that were of extremely limited distribution, in that they occurred only at one or two sites and had a high potential impact (e.g. *Ageratina riparia* (Asteraceae), *Ligustrum lucidum* (Oleaceae)). These weeds were accorded the highest priority for control, becoming the 'Class 1' weeds.

Species with limited distribution and slow spread came last on the priority list. Usually these plants have seeds that are not dispersed effectively, so while the species is a problem for the native vegetation in the immediate area, we expect it to remain in its current locality. There is still the danger that an apparent slow rate of spread is due to previous suppression by marsupial browsing. We continue to informally monitor the lower-priority species, to avoid being taken by surprise.

There are five invasive species that are distributed over the entire island: two species of pine (*Pinus radiata* and *P. pinaster*), *Ulex europaeus*, *Ageratina adenophora* (Asteraceae) and *Hakea sericea* (Proteaceae). This group was positioned in the middle of the priority list for control.

Following our initial categorisation, the priority ranking of invasive weeds was revised using a system of scoring each weed on its biological characteristics and potential impacts on the native plant communities of Rangitoto (Owen 1997; Wotherspoon and Wotherspoon 2001). There were no major alterations to the placement of weed species in the priority list as a result of this change in method, but the scoring system has advantages in clearly setting out the rationale behind the priority classes and forcing a complete and objective assessment of the weed flora.

The scores for each weed were used to group the weeds into three new priority classes. The Class 1 weeds, those with very limited distributions and serious impacts on the native systems, were drawn from the top half of the list, defined by their restricted distribution scores. The remainder of the top half of the list became Class 2, and the second half of the list became Class 3.

WEED MANAGEMENT STRATEGIES

Management units

The island was divided up into manageable sectors for weed control. These sectors were further subdivided into numbered plots as initial control work was undertaken. The plots were sized so that they could be searched in a day by a team of four, though during initial control the time taken to control the weeds often exceeded a day.

To counter the re-invasion potential from Motutapu Island, the south-western third of this island was designated a weed buffer zone. The buffer zone is large enough to include the high ridge closest to Rangitoto, hopefully catching most of the wind-borne weed seed, and we hope it encompasses most of the foraging activities of the local frugivorous birds. There was a suite of potential weeds growing in residents' gardens within this buffer zone, and six species of the 20 highest priority Class 1 weeds.

Management objectives

The life of the current phase of the Rangitoto Weed Control Plan takes us to 2006. There are two management objectives for each weed:

- a long-term management objective, based on what we considered achievable over approximately 15 years; and
- a five-year management objective, based on what we considered could be achieved over the five-year life of the plan, given the resources available.

The five-year management objectives were added to the plan recently. Weed control records from the previous five years are used to estimate the time and resources required to attain each goal. Both the five-year and long-term objectives will be reviewed in 2006, along with the rest of the weed management plan. Management objectives for each species are given in Table 1.

Eradication

Eradication is the goal for those weeds with a very limited distribution, for which the chances of unassisted re-introduction are effectively nil, and for which an effective control technique exists. Eradication is assumed to be complete when the control site has been cleared for a time period exceeding the known life of the seedbank. It is a difficult endpoint to define as the life of a seed in the seedbank can only ever be an estimate, and often nothing is known about the seedbank. We expect to have eradicated only three species by 2006, after 11 years of weed control (*Alocasia brisbanensis*, *Tradescantia fluminensis*, and *Spartium junceum*).

Control to zero density

Controlling to zero density involves maintaining a density of nil adult plants. It is the goal for those weeds with a limited distribution, but with either a very persistent (or unknown) seedbank or a strong likelihood of re-invasion from off the island. Control to zero density is the management objective for most of the weed species, considered achievable by 2006 for 35 species. We currently maintain 17 species at zero density (Table 1).

Sustained control

Sustained control aims to control the species to a defined density when it is unrealistic to maintain a nil density of adult plants, or when the species is only removed from situations in which it does the most damage, for example controlling gorse on open lava flows. This is the management objective used for weed species that are widespread across the island.

Control and eradication strategies

The strategy for control of the 72 weed species is to eradicate or control the class 1 species as a first priority then control species in classes 2 and 3 on a geographic basis - sector by sector. The decision was made early in the programme to focus control effort on the class 1 weeds - those weeds which have a high impact on the native vegetation, which were very limited in distribution, and have efficient seed dispersal mechanisms. Class 1 weed sites are visited and controlled every year. We have made good progress against these species, reducing many to zero density (Table 1), saving resources and time by controlling those problem weeds before the populations increased so much that control to zero density would not have been a realistic option.

An example of a weed species that in hindsight should have been controlled sooner is the vine *Dipogon lignosus*. This species occupied only a few square metres of forest canopy at the bach settlement of Rangitoto Wharf in 1990. By 1995 it had spread to cover almost one hectare of forest canopy in the same area, and had multiplied from a few vines to many hundreds. It is likely that this weed had been suppressed by possum and wallaby browsing which prevented it from establishing to a level where its invasive nature became apparent. While it is difficult to predict the responses of plants to major ecosystem change such as the removal of a browser, some control work around 1990 would have saved the many hours of mile-a-minute control work which is still going on in this area.

Weed control experience on Rangitoto has shown that all the weeds at a given site are best treated at the same time. Many of the weedy areas around bach communities have a ground tier dominated by the lower priority weeds *Nephrolepis cordifolia* (Davalliaceae) and *Crassula multicava*. Any area treated for a higher priority weed is rapidly invaded by these species and their dense growth obscures seedlings of higher priority weeds such as *Rhamnus* and *Asparagus asparagoides* (Liliaceae). Initial control of the class 1 weeds is now complete, and yearly follow-up control usually incorporates the lower priority weeds at the site.

The priority for control of sectors rests on the distribution of the more widely distributed, high priority class 2 weeds. The general strategy aims to slow the spread of weeds by controlling the least infested of the sectors first, working towards the most heavily infested areas. To contradict

Table 1 Invasive weeds of Rangitoto, grouped by control priority class. Management objective E=eradicate, Z=control to zero density, SC=sustained control.

Botanical name	Common name	Long-term management objective	Five-year management objective	Current status
Priority Class 1				
Acmena smithii	monkey apple	Z	Z	Z
Ageratina riparia	mistflower	Z	Z	
Alocasia brisbanenesis	elephant's ear	E	E	Z
Anredera cordifolia	madiera vine	Z	Z	
Berberis glaucocarpa	barberry	Z	Z	Z
Crataegus monogyna	hawthorn	Z	Z	Z
Ficus rubiginosa	Port Jackson fig	Z	Z	
Hakea salicifolia	willow-leaved hakea	Z	Z	
Hedera helix	ivy	Z	Z	Z
Iris foetidissima	stinking iris	Z	Z	
Jasminum polyanthum	jasmine	Z	Z	Z
Ligustrum lucidum	tree privet	Z	Z	
Ligustrum sinense	Chinese privet	Z	Z	
Lonicera japonica	Japanese honeysuckle	Z	Z	
Lycium ferocissimum	boxthorn	Z	Z	Z
Myoporum insulare	Australian ngaio	Z	Z	Z
Pennisetum clandestinum	kikuyu grass	Z	Z	Z
Phoenix canariensis	Canary Island palm	Z	Z	
Polygonum capitatum	pink headed knot weed	Z	Z	
Racosperma longifolium	Sydney golden wattle	Z	Z	Z
Rubus fruticosus	blackberry	Z	Z	
Senecio angulatus	cape ivy	Z	Z	Z
Spartium junceum	Spanish broom	E	E	Z
Tradescantia fluminensis	wandering jew	E	E	Z
Vinca major	periwinkle	Z	Z	Z
Priority Class 2				
Agapanthus praecox	agapanthus	Z	SC	
Agave americana	century plant	E	Z	
Ageratina adenophora	Mexican devil	SC	SC	
Araujia sericifera	moth plant	Z	SC	
Asparagus asparagoides	smilax	Z	SC	
Asparagus scandens	climbing asparagus	Z	SC	
Buddleia davidii	buddleia	Z	SC	
Chrysanthemoides monilifera	bone-seed	Z	SC	
Cortaderia jubata	purple pampas grass	SC	SC	
Cortaderia selloana	pampas grass	SC	SC	
Cotoneaster glaucophyllus	cotoneaster	Z	SC	
Crassula multicava	pitted crassula	E	SC	
Cymbalaria muralis	ivy-leaved toad flax	E	SC	
Dipogon lignosus	mile-a-minute vine	Z	Z	Z
Erica arborea	tree heath	Z	Z	Z
Erica lusitanica	Spanish heath	Z	Z	Z
Erigeron karvinskianus	Mexican daisy	Z	SC	
Gladiolus natalensis	wild gladiolus	Z	SC	
Hakea sericea	prickly hakea	SC	SC	
Hypericum androsaemum	tutsan	Z	Z	
Nephrolepis cordifolia	tuber ladder fern	Z	SC	
Paraserianthes lophantha	brush wattle	Z	SC	
Pinus spp.	pines	Z	SC	
Rhamnus alaternus	rhamnus	Z	SC	
Ulex europaeus	gorse	SC	SC	

Table 1 Continued. Management objective E=eradicate, Z=control to zero density, SC=sustained control.

Botanical name	Common name	Long-term management objective	Five-year management objective	Current status
Priority Class 3				
Aeonium x. *haworthii*	small pinwheel	Z	SC	
Aloe saponaria	soap aloe	Z	SC	
Bryophyllum diagremontianum	lizard plant	Z	SC	
Carica pubescens	mountain pawpaw	Z	SC	
Carpobrotus edulis	ice plant	Z	SC	
Centranthus ruber	valerian	Z	SC	
Crassula coccinea		Z	SC	
Chlorophytum chloronotum	spider plant	E	SC	
Epidendrum sinabaeum	crucifix orchid	Z	SC	
Eriobotrya japonica	loquat	Z	Z	
Gomphocarpus fruticosus	swan plant	Z	SC	
Impatiens sodenii	shrub balsam	Z	Z	
Lavandula dentata	lavender	Z	SC	
Lilium formosanum	lily	E	Z	
Maurandya erubescens	Maurandya vine	Z	Z	
Pelargonium spp.	pelargoniums	E	SC	
Polygala myrtifolia	sweet pea shrub	Z	SC	
Sedum mexicanum		E	Z	
Tecomaria capensis	cape honeysuckle	E	Z	
Watsonia bulbilifera	bulbil watsonia	Z	SC	
Watsonia meriana	watsonia	Z	Z	
Zantedeschia aethiopica	arum lily	Z	Z	

this however, the first sector to receive control work was the very weedy central scoria cone, at the summit. The decision was made to control this sector first because the scoria substrate supported the greatest number of Class 2 weeds of any sector, a number of which were suspected to be capable of spreading onto the broken lava surface of the rest of the island. It was also a relatively easy substrate to work on, and rapid progress was made. Until 1996 Spanish heath (*Erica lusitanica*) formed between 30 and 60% of the canopy on the northern side of the cone, and is now at zero density. Tutsan (*Hypericum androsaemum*) formed a solid ground cover of 6 ha on the south side of the cone, and seemed to be extending its range downhill from Wilson's Park onto the lava flows. It has been controlled every year since 1996, and is now occasional throughout this area.

Control of the most widespread of the Class 2 species, the pines and *Ulex europaeus*, has proved to be best undertaken using helicopters. Pines were mapped by Differential Global Positioning System (DGPS) from a helicopter and have proved relatively cheap to control, using forestry contractors and helicopter access.

The most abundant and highest scoring of the Class 2 weeds is *Rhamnus alaternus*, so the strategy for control by sectors is essentially dominated by the need to gain control of this weed.

Rhamnus alaternus

Rhamnus is a weed of great concern in all the forest types on Rangitoto. It is a hard-wooded, small (up to 10m) dioecious tree that is able to establish under a light canopy or in full sun (Fromont 1995). It reaches maturity in only three years, has a fast growth rate (height increase of up to 800 mm per year on Rangitoto), and a bird-dispersed seed.

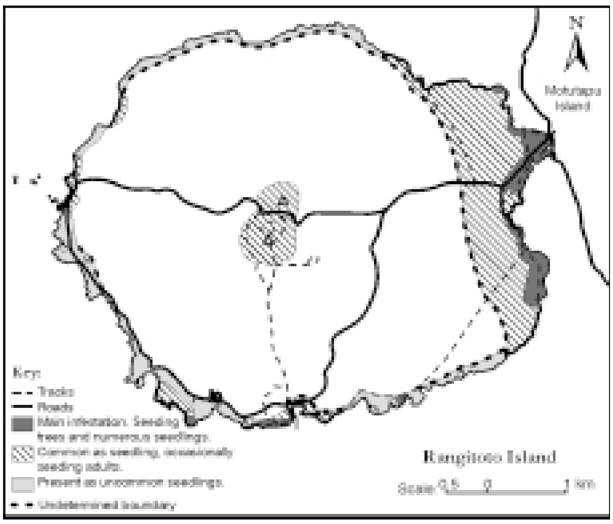

Fig. 2 Distribution of *Rhamnus alaternus* on Rangitoto Island.

Because its growth rate is faster than most native shrub and tree species, *Rhamnus* quickly dominates and extends islands of young vegetation on the lava flows. Without a substantial control programme for *Rhamnus* it is a fair expectation that it will spread across the remainder of Rangitoto, dominating and excluding pohutukawa from what are now open lava fields, outcompeting the subcanopy tiers in all forest types, and infiltrating the canopy of the lower scrubby vegetation types.

The extent of the *Rhamnus* infestation on Rangitoto can be seen in Fig. 2. While it is not present over the whole island, it is found all around the coast, and in the summit cone area. The main infestation comprises 187 hectares at Islington Bay and Gardiner Gap where Rangitoto and Motutapu Islands are joined (Fig. 2).

The control strategy aimed to control and maintain *Rhamnus* at zero density everywhere except for the main infestation at Islington Bay and Gardiner Gap before tackling the main infestation. We are very close to achieving this, and initial control of the main infestation began very recently by spraying *Rhamnus* and other weed species on the Motutapu cliffs from a helicopter.

Surveillance

A recent analysis of the weed invasion threat identified Motutapu Island, Auckland City, and North Shore City as the most likely contributors to the Rangitoto weed flora (Julian 1999). A surveillance programme is included as part of the weed management plan. Surveillance concentrates on the coast closest to Auckland and the North Shore to try to detect new invasions shortly after they arrive.

OPERATIONAL METHODS - THE EXECUTION OF THE PLAN

Searching for and controlling weeds is meticulous work, requiring diligence and keen observation. The Rangitoto weed team works in groups of three to five, systematically searching and controlling each plot side by side in a line. One end person follows a plot boundary and the other end person reels out biodegradable cotton from a hip chain, the cotton line indicating the position of the next swath. For each plot the amounts of control time and herbicide used on each species is recorded. Comparison of these data from year to year, when the same method is used, indicates the progressive reduction of each weed.

We use GPS technology extensively. The rough terrain and an absence of landmarks make navigation over Rangitoto difficult, so recent improvements in GPS technology have revolutionised the way we map and relocate weed infestations on the ground. All helicopter work is guided by GPS, and we have used helicopters to map weed infestations from the air, later relocating them for control from the ground.

Chemical control is necessary on Rangitoto simply because digging weeds out is impossible over most of the island. Many of the weeds, to the best of our knowledge, had never been subjected to control before. We conducted a range of control trials in an effort to find effective herbicides and application methods. These included foliar spraying, painting the trunks with herbicide, (sometimes chipping the bark off first), and felling the weeds and painting the stumps. The trials initially involved replications of treated individuals and sites, and are now incorporated into the general control plan as we continue to refine the successful methods.

DISCUSSION

The 'how-to' aspects of weed control are crucial to the development of a weed management plan. Operational details such as available control methods, logistics of transporting or accommodating staff, and resources available, will ultimately dictate the outcomes of weed management. Operational details are just as important as the biological information about each weed, ecological information about the systems being protected, and weed distribution patterns, when devising a plan.

Above all we have found flexibility in a weed management plan to be very important. During the five years over which the Rangitoto Weed Control Plan developed, there were additions to the weed list, changes to the priority classes, evolving control techniques, new herbicides on the market, and improving GPS technology. All invoked changes to the plan, in varying degrees. To retain the flexibility needed to take advantage of new information, our weed control planning is fine-tuned annually. A short annual plan allows for priorities to be temporarily swapped, and for resources to be redirected to take advantage of changing circumstances or to correct setbacks.

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Impacts and control of introduced small Indian mongoose on Amami Island, Japan

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Abstract Thirty individuals of the small Indian mongoose (*Herpestes javanicus*) were released on Amami Island, Japan in 1979 to control the venomous habu snake (*Trimeresurus flavoviridis*) and the black rat (*Rattus rattus*). However, the mongoose has had a major negative impact on agriculture and the native animals in mountainous areas instead of controlling snakes. A total of 3886 mongooses were trapped by pest control measures of the local government and an eradication project of the Environment Agency in the first year of the project (fiscal 2000). The population of the mongooses and annual growth rate were estimated at 10,000 individuals and 30% respectively before the eradication project. The project is in its early stages and there are many tasks to be addressed. Further eradication projects should take into consideration the low density and partial distributions of the mongoose population in mountainous areas.

Keywords small Indian mongoose; introduction; native animals; eradication; Amami Island.

INTRODUCTION

Amami Island is 710 km^2 in area and 694 m in maximum elevation, and 70% of the island is covered by forest. The island, one of the small islands of the Ryukyu Archipelago in the most south-western part of Japan, has many endemic and threatened species (Table 1). The habu, (*Trimeresurus flavoviridis*), a dangerously venomous crotalid snake, inhabits the higher elevations of the islands of the Ryukyu Archipelago, including Amami Island. The snake is feared by local residents because of the high frequency of encounters and severe consequences of its bite. The snake is encountered in fields during the day, along roads at night, and around residential areas. During the period from 1954 to 1998, 3600 persons were bitten and 50 persons were killed by the snake on Amami Island (Kagoshima Prefecture Office 1999).

Many measures to reduce incidences of snake bite and fatalities have been successively employed, including trapping, poisoning, alteration of habitat around housing, and serum development. Pest control by biological means was also employed to reduce the snake population and their principal prey, the black rat (*Rattus rattus*). Before releasing the small Indian mongoose (*Herpestes javanicus*), 871 individuals of the Japanese weasel *Mustela itatsi*, were introduced as a snake and rat predator on Amami Island during 1954-1958, but none remain. More than 2000 weasels were released on the other eight small islands of the Amami archipelago in the same period. They did not colonise on six islands (including Amami Island) occupied by snakes, but colonised successfully on three islands where there are no snakes. This is thought to be due to competition for the same prey as snakes and also predation by snakes because they share the same nocturnal activity (Hayashi 1979). In contrast, the small Indian mongoose

Table 1 Threatened native species (assessed using IUCN categories) on Amami Island and other islands that have been recorded in the diet of mongoose. Identified mongoose food items are marked with an asterisk.

Category	Mammals	Birds	Reptiles	Amphibians
Critically Endangered		*Zoothera dauma major* [1,2]		
Endangered	*Crocidura orii* [2] *Diplothrix legatus* [1]* *Tokudaia osimensis* [1]* *Pentalagus furnessi* [1]*	*Scolopax mira** *Dendrocopos leucotos owstoni* [1,2]		*Rana ishikawae* [2?]
Vulnerable		*Dendrocopos kizuki amamii* *Erithacus komadori* [1]* *Garrulus lidthi* [1,2]*	*Japalura polygonata** *Eumeces barbouri**	*Tylototriton andersoni* *Rana amamiensis* *Rana subaspera* [2]
Lower Risk	*Crocidura horsfieldii watasei**	*Columba janthina janthina* [1]	*Achalinus werneri* *Calliophis japonicus japonicus* [2]*	*Cynops ensicauda*
Other native species			*Cyclophiops semicarinatus**	

[1] Japanese Natural Monument. [2] Endemic or main population on Amami Island.
Food species were cited from Environment Agency (1999) and Yamada *et al.* (2000).

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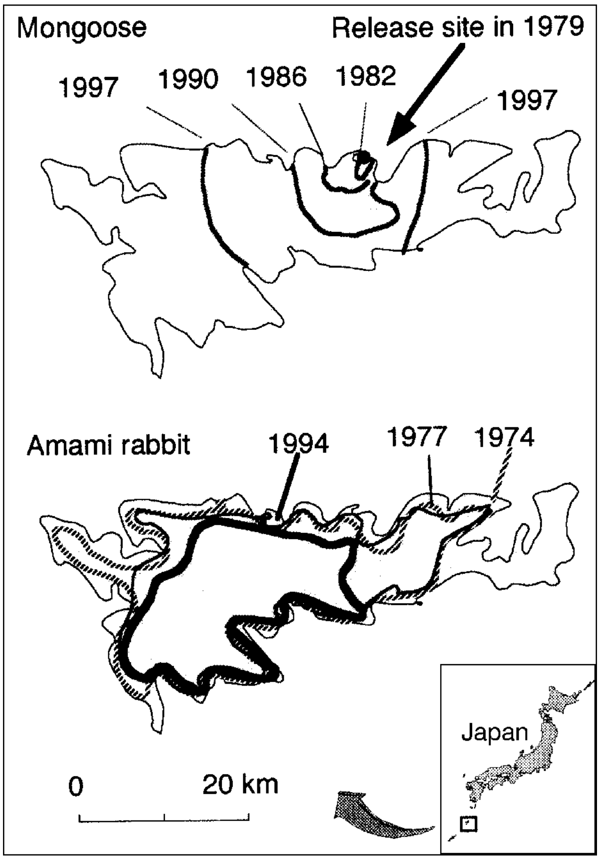

Fig. 1 Release site of the small Indian mongoose (*Herpestes javanicus)* at Naze City in 1979 and current distributions of mongoose (Environment Agency 1999), and Amami rabbit (*Pentalagus furnessi*) (Sugimura *et al.* 2000).

has successfully colonised Amami Island. In Japan, the mongoose had already successfully colonised Okinawa Island from 1910 (Kishida 1931). According to mtDNA analysis, the original individuals of mongooses on Amami Island are thought to have been brought from Okinawa Island (Sekiguchi *et al.* 2001).

This paper reviews and assesses the impacts of the mongoose and control practices on Amami Island.

Release and colonisation

Thirty mongooses are believed to have been released to control snakes around a new public educational facility opened in a forested suburb of Naze City on Amami Island in 1979 (Fig. 1). However, there is no official record of the release. Since then, the mongoose has been expanding its distribution from the release site, covering a 10 km radius by 1989 and a 20 km radius by 1997, covering half of mountainous areas occupied by many threatened species, such as the Amami rabbit (*Pentalagus furnessi*). The rate of range extension was estimated as 1 km per year. After 20 years the population size was estimated at 5000-10,000 mongooses in 1999 (Environment Agency 1999).

Agricultural impacts

The mongoose has a large impact on crops (taro, sweet potato, melon, watermelon, loquat, etc.) and poultry in farmland. The economic cost of the damage rapidly increased in 1994 (USD7000), 1995 (USD32,000), 1996 (USD64,000), 1997 (USD110,000), 1998 (USD100,000) and 1999 (USD80,000). Some farmers trapped mongooses to protect crops on their farmland before 1993 when the local government began to control the mongoose.

Predation damage on endemic animals

Since advancing into mountainous areas in around 1986, the mongoose has had a predatory impact on the native animals in the mountainous areas, as listed in Table 1. However, there was almost no evidence of predation of snake by mongoose (Abe *et al.* 1999; Environment Agency 1999; Yamada *et al.* 2000). According to our findings, insects (40%), other invertebrates (90%), amphibians and reptiles (60%), mammals (20%), and birds (15%) were observed in 89 pellets of mongoose collected in the habitat of the Amami rabbit (Yamada *et al.* 2000). Eight percent of pellets contained the Amami rabbit (Fig. 2). Although the mongoose chiefly preyed on insects and birds in all seasons, it tended to prey more frequently on amphibians and reptiles in summer and on mammals in winter. The distribution and abundance of the Amami rabbit are thought to have been reduced by the mongoose, as well as by habitat reduction due to forest cutting and infrastructure construction (Fig. 1; Sugimura *et al.* 2000).

Mongoose control

In the 1980s local scientists on Amami Island carried out ecological studies of mongoose populations, mostly near the release areas (Abe *et al.* 1999). The local government began to trap the mongoose in order to reduce crop damage in farmlands around the city from 1993 and the Yamato

Fig. 2 A small Indian mongoose *(Herpestes javanicus)* and inset a mongoose pellet containing Amami rabbit (*Pentalagus furnessi).*

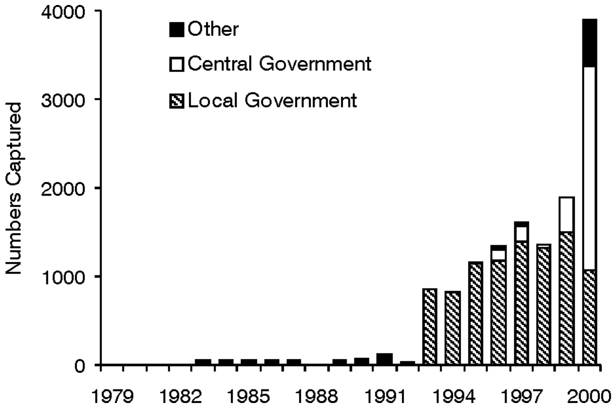

Fig. 3 Annual numbers of the small Indian mongoose (*Herpestes javanicus)* captured by traps on Amami Island (Environment Agency unpublished).

Village Office also began trapping from 1995. As many as 1100-1500 mongooses were captured by 15-20 trappers using 10-30 traps per person for seven to nine months (May-March) in a year (Fig. 3). Approximately 60-80% of those mongooses were captured by four to five skilful trappers.

After recognising the predation impact of the mongoose on threatened animals on the island, the Environment Agency of the Central Government carried out preliminary investigations during 1996-1999 into the possibility of eradicating mongooses from the whole island. The investigations assessed the following: expansion of the distribution, reproduction, food habits, estimation of population size and annual growth rate (5000-10,000 individuals in 1999 and 30%), and techniques for eradication by cage traps and wooden box traps using fish sausage as bait. Following on from the pilot investigations, the Environment Agency decided to begin a full-scale project to eradicate the mongoose from the whole island from 2000 by two methods: (1) great reduction of population using many traps during a short period (3 years) over the whole island (the annual target reduction was 4000-5000, including the number by pest control around farmland by the local governments during May to March, and by trapping in mountainous areas by the Environment Agency during October to March), and (2) long-term eradication until the species becomes extinct.

Results so far

A total of 3886 mongooses were captured by trapping, but the number was lower (87%) than the target number (4500) in the first year (fiscal 2000). The catch comprised 1073 animals by pest control and 2813 by the eradication project of the Environment Agency (Fig. 3). The small drop in numbers was caused by a one-third reduction in the number caught in a normal year by pest control and to the 10% reduction in the target number of the eradication project of the Environment Agency. Twelve to 22 trappers were engaged in the eradication project by the Environment Agency in the first year (October 2000-March 2001). Five of them worked for both the pest control and the eradication project. Most of the capture places and total number of traps they used, and total number of days for trapping were not reported accurately by themselves, because they do not usually record such data. Many of the mongoose captured around farmland were reported to the office as having been captured in mountainous areas. Capturing in farmland is more efficient than in mountainous areas because the density of mongooses in farmland areas is higher than that in mountainous areas.

Trappers were paid USD18 per mongoose by either the pest control or the eradication project upon taking the tail of a mongoose to either office. A total of USD50,000 was spent directly on the island by both agencies in the first year. However, some trappers lost the incentive to trap because of the reduction of capture efficiency as the number of mongoose presumably decreased.

Problems and conclusion

The project is in its early stages and there are many tasks to be addressed. Although there have been a few recent studies on controlling mongooses by chemicals in Hawaii (Smith *et al*. 2000) and on management implications on Mauritius (Roy *et al*. 2002), there are few successful examples of mongoose eradication in the world (Simberloff 2001). This is the first trial of eradicating mongoose. But there are many difficulties, including 70,000 people living mainly along the coast, many endangered animals in the mountain forests, and the venomous snake, Habu, on the large mountainous island. Therefore, even after this trial, it will be necessary to ensure: unified management by both the pest control and the eradication project for establishing year-round trapping and a strategy of eradication; monitoring the efficiency of eradication; introduction and development of more effective techniques; monitoring the effects on recovering the endemic animals and on the ecosystems including rat control; and exchange of information with experts in foreign countries. The continuous supply of more budget, manpower, public information, and research work will also be necessary.

Further projects must consider how to eradicate mongooses of patchy distribution and low density in mountainous areas. Most endemic animals on the island, including the Amami rabbit, seem to be vulnerable to this exotic predator because of their long isolation in an insular environment which lacked such a large active predator as the mongoose. For the conservation of the Amami rabbit and other native animals on Amami Island, more effective measures are needed to eradicate this invasive predator.

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It's often better to eradicate, but can we eradicate better?

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Abstract Invasive species eradications have achieved important conservation gains the world over. Growing numbers of eradications take place, however, in complex and highly altered ecosystems with high risks of unexpected ecological effects. Ecosystems that contain multiple invaders, have lost one or more native species along with their functional roles, or have undergone long-term change to soil and other site conditions can respond to eradication with mixed results. The most common secondary outcome of a single-species eradication is the ecological release of a second (plant or prey) exotic species previously controlled by the removed species (herbivore or predator) through top-down regulation. Examples of a variety of other undesirable secondary outcomes also exist, challenging invasive species managers to develop tools for predicting and averting these "surprises." Most unexpected outcomes can be understood and anticipated through knowledge about species interactions and the general ecological rules that they follow. Several tools that already exist, including thorough pre- and post-eradication monitoring and restoration measures such as re-seeding, simply need to be applied more routinely in eradication projects. Other areas deserve to be carefully explored, such as formal but qualitative approaches to ecological assessment during the planning stages of an eradication project. As eradication moves from narrow invasive species management to actively pursuing and practicing restoration, it will be able to achieve clear conservation results in increasingly challenging settings without accidental, adverse effects.

Keywords Invasive species eradication; secondary effects; species interactions; food web; ecological release; restoration.

INTRODUCTION

Invasive species now pose an enormous threat to the world's biological diversity, second only to land-use change (Chapin *et al*. 2000). Several global trends – growing human populations, transport, and tourism, the weakening of trade barriers as trade volumes skyrocket; ongoing habitat loss; and climate and atmospheric changes – will likely increase the movement, establishment, and spread of exotics (Mooney and Hobbs 2000). If biological invasions go on unabated, crude estimates predict the eventual loss of at least 30-35% of the world's species (McKinney 1998).

We have an opportunity to overcome this bleak vision with a steadily growing arsenal of knowledge, tools, and techniques for preventing and undoing biological invasions and their harmful effects. The case studies in this volume document the latest advances in undoing biological invasions in critical areas for biodiversity conservation. Many of these cases illustrate that a range of invasive taxa, including vertebrate animals, plants and insects, can be eradicated from a diversity of regions around the world (e.g. Veitch 1974; Allwood *et al.* 2002; Burbidge and Morris 2002; Coulston 2002; Dixon *et al*. 2002; Flint and Rehkemper 2002). The conservation potential of the projects described in this volume is especially great because they focus on islands, which contain a disproportionate share of the world's unique species (Whittaker 1998) and are especially vulnerable to the impacts of invasions (Atkinson 1989; Simberloff 1995). These case studies illustrate that island invasive species eradications are already an important and effective way to protect native biota and ecosystems.

These case studies also present an opportunity to learn from experience. Eradications take place in increasingly complex ecological contexts – in settings affected by multiple invaders (e.g. Algar *et al*. 2002; Bullock *et al*. 2002; Carter and Bright 2002; Coulston 2002; Klinger *et al.* 2002; Micol and Jouventin 2002; Mowbray 2002; Roy 2002; Rippey *et al.* 2002; West 2002), long-term damage to native populations and ecosystem function (e.g. Brown and Sherley 2002), and other global environmental stresses such as climate change (IPCC 2001). These complexities mean that restoring native systems is not always as straightforward as removing an invader. They also mean that eradications are more likely to have unexpected, undesirable effects, such as the accidental release of other exotic populations (Zavaleta *et al*. 2001).

What can go wrong?

To some extent, eradications will always be single, unreplicated experiments, so there will always be some surprise outcomes (Simberloff 1995, 2002). My goal is to help reduce undesirable outcomes of eradications through an assessment of why they occur and how they can be prevented. Eradications fail for a variety of reasons, including non-target impacts of the eradication method itself (e.g. Morris 2002; Torr 2002) and failure to eliminate the target organism (e.g. Varnham *et al.* 2002; Hammond and Cooper 2002; Burbidge and Morris 2002; Lovegrove *et al*. 2002; Bell 2002; Parkes *et al.* 2002). Other authors provide excellent critical overviews of how to avoid these types of problems (Moro 2002; Burbidge and Morris 2002). Here, I focus on the problem of unwanted, secondary ecological consequences of *successful* eradications – releases of other exotic populations, declines in native populations following eradication, and the failure of na-

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tive biota and ecosystems to recover once target invaders have been removed. Aspects of this topic have been discussed elsewhere (Zavaleta *et al.* 2001); in this paper I discuss some specific, possible solutions to unwanted secondary impacts.

Species interactions – both among exotics and between exotic and native species – lie at the root of most of these post-eradication outcomes in these categories. In invaded ecosystems, exotic species interact with each other and with native species largely according to the same rules that govern all species interactions. In any ecosystem, populations of producers, consumers, and predators are in part controlled by one another through food web and other biotic interactions, including competition and provision of habitat (Hairston *et al.* 1969; Fretwell 1987; Polis and Strong 1996). Every invaded ecosystem is unique in some way, but every invaded ecosystem also follows, at least qualitatively, the same set of basic rules that all ecosystems do. With these basic ecological rules in mind, managers and eradication experts can make great gains towards anticipating, planning for, preventing, and mitigating the unexpected.

The types of species interactions that produce undesirable eradication outcomes can be viewed as falling into three classes. The first and largest includes trophic (food-web) and competitive interactions, both between exotics and natives and among exotic species themselves. Both competition and trophic interactions are large categories of species interactions important to eradication outcomes, but they are necessarily linked in many cases. For example, eradication of feral pigs (*Sus scrofa*) and sheep (*Ovis aries*) in Hawai'i removed herbivores that controlled exotic plant populations (food-web interaction). Competition between exotic and native plants became a more important structuring force in the absence of top-down control by feral herbivores, with mixed results (Scowcroft and Conrad 1992). The other two, smaller classes of species interactions – provision of habitat by one species for another, and indirect interactions through the alteration by one species of site conditions for another – are discussed near the end of this section.

Trophic and competitive interactions

The rules governing food-web interactions and their relative importance in different ecosystems have long been studied and debated. Research in a range of ecosystems has shown that both bottom-up and top-down regulation of populations of consumers and producers can play important roles (Pace and Cole 1996; Pace *et al*. 1999; Polis 1999; Terborgh *et al*. 1999). The importance of these forces has implications for interactions in invaded ecosystems. Bottom-up regulation of predators by prey (Polis 1999) implies that, among other things, removing an exotic prey species could reduce both exotic and native predator populations. On Santa Cruz Island, California, USA, ecologists anticipate that feral pig (*Sus scrofa*) eradication will reduce native golden eagle (*Aquila chyrsaetos*) populations that prey on the pigs (Roemer *et al.* 2002). In

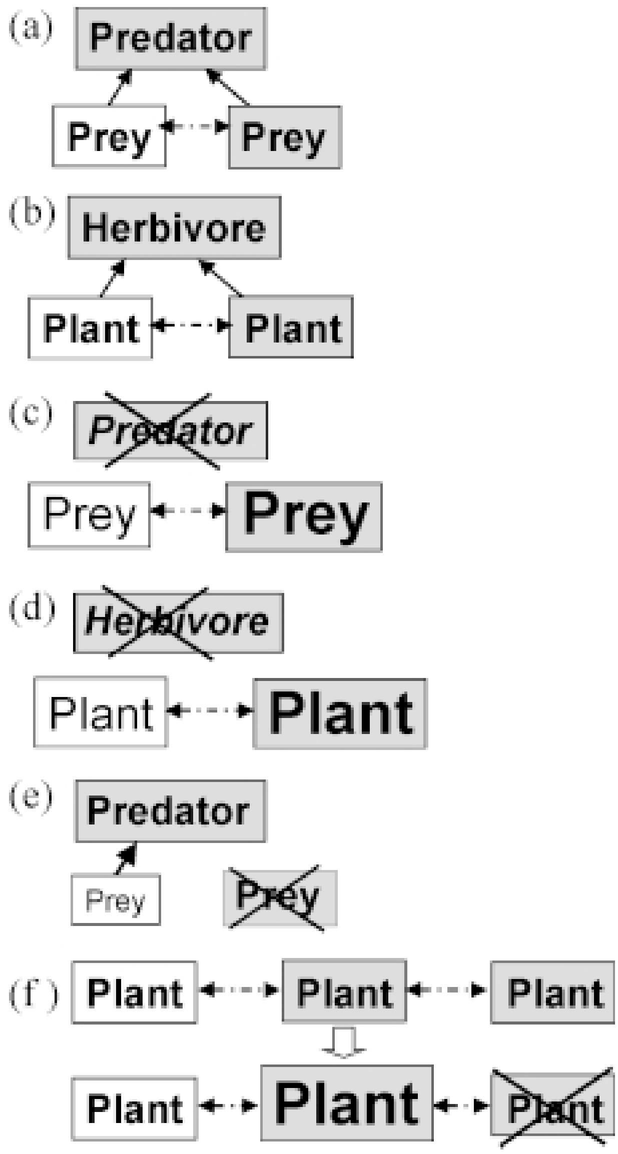

Fig. 1 Ecological release following removal of an exotic species. Grey boxes are exotics, white boxes are natives, dashed arrows indicate competition, and solid arrows point to consumers from the organism consumed. Font size indicates population size. In systems where (a) exotic predators consume both native and exotic prey or, (b) exotic herbivores consume both native and exotic plants, removal of the top consumer can lead to release of exotic (c) prey or (d) plant populations that can outcompete their native counterparts. In (e), removal of an exotic prey species from situation (a) leads an exotic predator to consume more native prey, a phenomenon known as prey-switching. In (f), removal of one exotic plant species leads to ecological release of a second exotic plant species, with no benefit to native plant populations. This phenomenon has been termed "the Sisyphus effect" by Mack and Lonsdale (2002). Figure adapted from Zavaleta *et al*. 2001.

this case, the reduction of the native raptor will be welcome; predation by pig-inflated golden eagle populations appears responsible for sharp reductions in endemic island fox (*Urocyon littoralis*) populations.

Similarly, top-down regulation of prey by predators (including regulation of plant populations by herbivores) (Terborgh *et al.* 1999) implies that removing exotic predators can increase populations of both native and exotic prey (Fig. 1a-d).

This kind of ecological release – of exotic prey or plants previously consumed by an introduced animal that gets removed – has occurred in a range of settings involving a range of exotic species (Fig. 1a-d). In some cases, an exotic predator controls populations of exotic prey species until the predator is removed. Mesopredator release, the rapid expansion of a prey population once top-down control by a predator has disappeared, could lead to negative effects if the expanded prey population competes with or consumes native biota. Eradications of feral cats (*Felis catus*) in the Orongorongo Valley, New Zealand (Fitzgerald 1988) and on Isabela Island, Mexico (C. Rodriguez, unpub. data) have led to increased populations of introduced rats. Merton *et al*. (2002) describe an explosive irruption of exotic crazy ant (*Anoplolepis gracilipes*) populations following, and possibly resulting from, the removal of rats from Bird Island in the Seychelles. Certain common invaders are known to feed on other exotic animals in a variety of settings. Data from Fitzgerald (1988) indicate that where introduced rabbits are absent, exotic rats generally make up more than two thirds of the diet of introduced cats on several islands where rats and cats co-occur (Fitzgerald 1988; Fitzgerald *et al.* 1991) (Table 1). On islands where introduced cats, rats, and rabbits all co-occur, rats make up a much smaller part of cats' diets – suggesting that in these settings, cats might be eating many rabbits instead of rats.

Table 1 Exotic rats in the diet of introduced cats on islands. Data from Fitzgerald (1988).

Islands without introduced rabbits	Occurrence of rats in diet (%)
Galapagos: Isabela	73
Galapagos: Santa Cruz	88
Lord Howe	87
Raoul	86
Little Barrier	39
Stewart	93
Campbell	95
Islands with introduced rabbits	**Occurrence of rats in diet (%)**
Gran Canaria	4
Te Wharau, NZ	3
Kourarau, NZ	Trace
Orongorongo, NZ	50
Mackenzie, NZ	2
Kerguelen	0
Macquarie	3

NZ=New Zealand
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Mesopredator release can potentially lead to cascading changes in entire ecosystems. On subantarctic Marion Island, pre-eradication studies found that feral cats fed heavily on exotic house mice (*Mus musculus*). The mice, in turn, ate large numbers of an endemic moth, *Pringleophaga marioni*, important to nutrient cycling on Marion (Bloomer and Bester 1990, 1992; Crafford 1990). Cat eradication could have released mouse populations, which in turn could have reduced moth abundance and subsequently changed patterns of soil nutrient availability.

Even more frequently, removal of an exotic herbivore releases populations of exotic plants from top-down control. Many islands have large numbers of exotic plants on them in addition to the more often focused-on exotic herbivores (e.g. Frenot *et al.* 2001). Bullock *et al*. (2002) describe how rabbit eradication from Round Island, Mauritius has increased plant biomass, but mainly by increasing the dominance of exotic species in the island's flora like *Chloris barbata* (North *et al.* 1994). Klinger et al. (1994, 2002) describe a similar outcome following the removal of sheep from Santa Cruz Island, U.S.A. On Santa Cruz vegetation cover has increased, but certain endemic plants species have declined, and exotic plants have proliferated in areas formerly grazed by the sheep. On nearby Santa Catalina Island, the removal of feral pigs and goats has increased plant diversity and vegetation cover, reducing potential for further topsoil erosion (Schuyler *et al.* 2002). However, exotic species contributed much of the gain in plant diversity and increased in both absolute and relative cover (Laughrin *et al.* 1994).

Only one exotic plant need be present in an ecosystem to pose a threat. The most dramatic exotic plant release described in this volume (Kessler 2002) involved a single species whose presence was unknown prior to exotic mammal eradication. Following removal of feral goats and pigs from Sarigan Island in the Commonwealth of the Northern Mariana Islands, the exotic vine *Operculina ventricosa* rapidly became superabundant. It now covers much of the island, but its effects on ongoing regeneration of the island's native forests and fauna remain unclear. On San Cristobal Island in the Galapagos, removal of feral cattle from areas containing suppressed populations of exotic guava (*Psidium guajava*) led to rapid development of dense, mature guava thickets (Eckhardt 1972). In a case like San Cristobal, herbivore removal can create a situation that for practical purposes may be irreversible. Browsers and grazers will consume guava seedlings and damage saplings, but they cannot reduce numbers of established, woody guavas once succession to these exotics has been allowed to progress.

In every one of these cases, successful eradication removed a damaging exotic from a threatened ecosystem. These cases make clear, though, that greater conservation gains are possible if these initial eradications are viewed as only first steps in a larger process of island restoration. In some settings with multiple invasions, ecological releases of other exotics can be anticipated, and steps can be taken to head off potential problems before, during, and after eradication. In others, unanticipated releases can be caught and managed effectively through a combination of contingency planning for surprise outcomes and post-eradication monitoring.

Diet switching and competition

Eradication of exotic prey without the simultaneous removal of introduced predators can also spell trouble when these predators are forced to switch their diets to native prey species (Fig. 1e). In New Zealand, introduced stoats (*Mustela erminea*) feed largely on introduced rats (*Rattus rattus*) and common brushtail possums (*Trichosurus vulpecula*) (Murphy and Bradfield 1992; Murphy *et al*. 1998). Efforts to reduce all three of these species together resulted in successful control of the rats and possums, but not the stoats. With the exotic prey species much reduced, the remaining stoats switched their diets to include more native birds and eggs. This type of prey switching, under the wrong circumstances, could potentially extirpate a native prey species in an island setting. Since removing the exotic predator first could lead to increased exotic prey abundance (Fig 1. a,c), whether to remove exotic predator or prey first can pose a serious quandary. The solution to this scenario depends on, among other things, the feasibility of a successful dual eradication; the ability of native prey populations to withstand temporary increases in predation; and the increased difficulty of successful prey eradication that would result from an exotic prey population expansion following predator removal.

Competition plays important roles in the responses of multiply-invaded ecosystems to eradications, both in concert with trophic links and on its own. When an exotic herbivore is removed from a multiply-invaded island, top-down control may cease to be the main force suppressing both exotic and native plant populations. In the new, herbivore-free setting, competition between native and exotic plants might play a bigger role in shaping who "wins." Invasive exotic plants often have life history traits such as large and frequent seed crops and short times to reproductive maturity (Mack 1996; Rejmanek and Richardson 1996). These can provide a competitive advantage over island natives and endemics, so that the winners of these "contests" at least in the short term are, unfortunately, often the exotics. Variations on this shift from top-down to competition-driven threats posed by exotics have followed pig, sheep, goat, rabbit, and other herbivore eradications on islands around the world, in the Channel Islands U.S.A, Mauritius, Oceania, the Galapagos, Hawaii, and Mexico, among other locations.

The removal or control of a single exotic plant species from an ecosystem containing multiple exotic plants can also produce competition-mediated releases, with discouraging results. Mack and Lonsdale (2002) provide several examples of exotic plant removals on land and in aquatic systems that appear to have led to increases of other exotic plant populations released from competition, with no clear benefits to native biota. Exotic plant removal may achieve desirable results only if all invasive species present are targeted together, or if native plants are actively restored to prevent other exotics from grabbing resources freed by the removal.

Habitat and indirect interactions

The second class of interaction that can complicate eradication planning and execution is a positive association between a native and an exotic species. Elsewhere in this volume, Carter and Bright (2002) describe how exotic but non-invasive Japanese red cedar (*Cryptomeria japonica*) plantations on the island of Mauritius provide refuges for native birds from introduced predatory macaques (*Macaca fascicularis*). In a case like this, removal of an exotic species (Japanese red cedar) would indirectly increase the impacts of another exotic species on endemics with high conservation value. In the western U.S.A, large areas of invasive saltcedar (*Tamarix ramosissima*) trees have replaced the historical riparian forest habitat of the endangered south-western willow flycatcher (*Empidonax trailii* var. *extimus*) (USFWS 1997). In these areas, the flycatcher now depends on the invasive saltcedar as nesting habitat. Large-scale removal of these saltcedar stands without accompanying native forest restoration could, some government officials argue, threaten the endangered songbird. Saltcedar control within the range of the flycatcher will likely need to include careful planning and restoration measures to meet the requirements of the U.S. Endangered Species Act.

A third class of species interaction, which can create a need for significant post-eradication restoration work, is an indirect negative effect of an exotic on native species that persists after the removal of the exotic. The clearest examples of this type of interaction involve exotic plants that alter site properties. Invasive iceplant (*Mesembryanthemum crystallinum*) salinises soils so much that native vegetation may not be able to recolonise after its removal (El-Ghareeb 1991; Vivrette and Muller 1977). Restoration of iceplant-invaded areas on Santa Barbara Island, Channel Islands National Park, U.S.A is expected to require substantial soil restoration measures beyond the removal of the exotic plant (Philbrick 1972; Halvorson 1994). Similarly, invasive trees and shrubs of the genus *Tamarix* in the south-western United States salinise streamside soils to levels not tolerated by many native organisms (Jackson *et al*. 1990; Busch and Smith 1995; Shafroth *et al*. 1995; Wiesenborn 1996). Nitrogen-fixing plants, such as invasive Scotch broom (*Cytisus scoparius*) and French broom (*Genista monspessulana*) in coastal California, U.S.A, can increase soil nitrogen availability over time (Bossard *et al*. 2000). When these species are removed from long-invaded sites, this high nutrient availability can increase site susceptibility to re-invasion by exotic annuals (K. Haubensak pers. comm.). In cases like these, altered site conditions might recover over time without intervention. Leaving these kinds of sites to recover on their own, though, can come at cost. Soil erosion, susceptibility to re-invasion, and an absence of forage and habitat for native animals all could create bigger and more costly management challenges than pursuing active site restoration from the start.

With eradications taking place in increasingly complex and altered settings, a wide range of unexpected outcomes are possible (Table 2). Some of these potential outcomes are less likely than others because the particular conditions required to produce them are rare, such as the case of a predator removal releasing exotic plant populations through cascading changes in ecosystem interaction webs. Others, such as the failure of a reduced or extinct native population to recover, or the ecological release of an exotic competitor or prey species, occur with undeniable regularity. Are these "side-effects" of well-intentioned eradications just noise around overwhelmingly successful conservation projects, or can they completely undo the good intentions of an eradication project and create even more serious problems? The answer is probably both, depending on context and on the steps taken to cope with them before and while they occur.

What we can do

Removing exotic species from ecosystems is rarely an end in itself. The ultimate goal of most eradications should be to restore the diversity and functioning of native ecosystems (but see Browns Island case (Veitch 2002)). Most practitioners now recognise this objective, so narrow definition of the goals of eradication is not really a problem. For instance, nearly every eradication case study in this volume specifies its goal in terms of allowing recovery, protecting native species, restoring biological diversity, or some other aspect of conservation.

What fewer of these case studies describe is an active pursuit of their conservation goals, through specific steps like restoration planning or monitoring. This may be partly because the focus of this volume is the process of eradication itself. Still, fewer than half of the case studies in this volume mention any pre- or post-eradication monitoring other than search for missed target individuals and immediate non-target effects. Given the explicit conservation and restoration goals of most eradication projects, this is surprising. Without pre-eradication evaluation of a project's context, managers cannot reliably avert or plan for the undesired side effects of eradication in a complex setting. Without at least some post-eradication monitoring, managers cannot possibly catch totally unanticipated side effects or know whether and when to implement contingency plans for dealing with undesired outcomes.

Table 2 Potential, undesired effects of exotic species removals. Removals of exotic plants, herbivores, and predators (top row) can alter interactions with other exotic and native species in an ecosystem (left columns) in ways that move the system further from a desired state (table cells). Exotic species removals can also be insufficient to permit natural ecosystem recovery, as when exotic plants have rendered site conditions inappropriate for native establishment. See text for further discussion. Bolded statements indicate outcomes documented by at least one case study discussed in this paper or elsewhere.

Species affected		Exotic species removed		
		Plant	Herbivore	Predator
Plant	exotic	**Competitive release**	**Top-down release**	Cascading release
	native	**Site alteration prevents recovery**	**Small population prevents recovery**	Loss of dispersal vector
Herbivore	exotic	Food switching to native plant	Competitive release	**Top-down release**
	native	**Loss of protection/habitat**	Small population prevents recovery	**Small population prevents recovery**
Predator	exotic	**Thrive in native vegetation**	**Switch to native prey**	Competitive release
	native	Loss of protection/habitat	Decline due to absence of prey	

Leveraging information to guide eradication

Without post-eradication follow-up, eradication experts as a community also cannot accumulate valuable knowledge about project outcomes. Cromarty *et al.* (2002) identify a need not only to define long-term restoration goals, but also to better understand the downstream effects of removing exotics. We need both of these pieces: only by understanding downstream effects can we determine how to meet those long-term goals, and only specific, defined long-term goals can consistently guide decisions that produce (mostly) the "right" downstream effects and not (as many of) the "wrong" ones. In the long run, knowledge accumulated through consistent follow-up monitoring is a major way for the global eradication community to improve and refine its techniques and to communicate the importance of invasive species removals to new and sometimes sceptical audiences.

While avoiding surprise outcomes and improving eradication techniques require understanding the ecological systems where eradications take place, this understanding has to come with the recognition that many islands are, to varying degrees, in states of crisis. There are costs to waiting for information to be gathered. As much as possible, research needs to be incorporated into actual conservation projects. Short-term, pre-eradication studies can provide useful insights into potential ecosystem responses to an invasive species removal. For example, careful, pre-eradication food trial experiments to quantify the plant food preferences of introduced rabbits on islands have qualitatively predicted plant community responses to rabbit eradication on small, simple islands with very few plant species and no other exotic herbivores (Donlan 2000). These kinds of studies may not work as well, though, in the complex settings where predictive tools are most needed. The same food preference trials yielded little information on islands with modestly diverse (<50 species) floras and multiple exotic herbivores (E. Zavaleta and B. Tershy, unpub. data) (Fig. 2). Constructing exclosures while exotic herbivores are still present can also provide a window into how vegetation might respond to herbivore removal in more complex settings. Interannual and spatial variability and time lags in community response, however, all limit the ability of one or a few years of exclosure data to predict an entire island's response over decades.

Fig. 2 A feral rabbit (*Oryctolagus cuniculus*) selects an exotic forb (*Tribulus cistoides*) over an endemic bunchgrass (*Aristida pansa*) in a food preference trial on Clarion Island, Mexico.

Within the planning of any given eradication, then, a better alternative to collecting large quantities of information in search of clear answers might be to identify the *minimum* information necessary to suggest wise decisions. Models exist for how to both identify and use minimum necessary information in this way. Qualitative assessment methods, such as decision trees (Reichard and Hamilton 1997) and rule-based models (Starfield *et al*. 1989; Starfield 1990) allow one to characterise a species or a whole system with little or no quantitative information. For example, the North American woody invaders decision tree of Reichard and Hamilton (1997) allows one to assess whether any woody species is safe to import based on yes/no answers to two to seven questions about its basic ecology. It should also be possible to improve decisions about island eradication planning with a qualitative understanding of key aspects of the island's condition and ecology. Basic knowledge of the exotic species present in a system, the likelihood for interactions among them and with native species, and the extent of damage they have caused can flag areas to consider more carefully in eradication planning. Figure 3 provides a rough example of what such a planning guide could look like for island eradications. It starts with three qualitative questions about the ecology of the island on which eradication is to take place:

- Is there (or could there be) more than one exotic species on the island?
- Has the target species eliminated or greatly reduced any native populations on the island?
- Has the target species altered site conditions in any long-term way, such as severe soil erosion or salinisation?

If the answer to any of these questions is "yes," planners could consider additional questions about the eradication and restoration process. Perhaps the most critical aspect of this process is the evaluation of tradeoffs before taking action: what are the worst-scenario costs of proceeding with no further planning or information? And what are the worst-scenario costs of waiting? In some cases the best strategy for avoiding disaster, such as an extinction, may still be to proceed with immediate eradication. In other situations, the best strategy may call for adding "surveillance" steps, such as post-eradication monitoring to catch unwanted changes early, or "action" steps, such as native species re-seeding/re-introduction in conjunction with exotic species removal (see Zavaleta *et al*. 2001) or simultaneous removal of more than one species (Murphy *et al.* 1998).

Often, the single best strategy, from a holistic conservation standpoint, will not be obvious because outcomes cannot be fully predicted. On Clarion Island in the Revillagigedo Archipelago, Mexico, exotic rabbit, sheep,

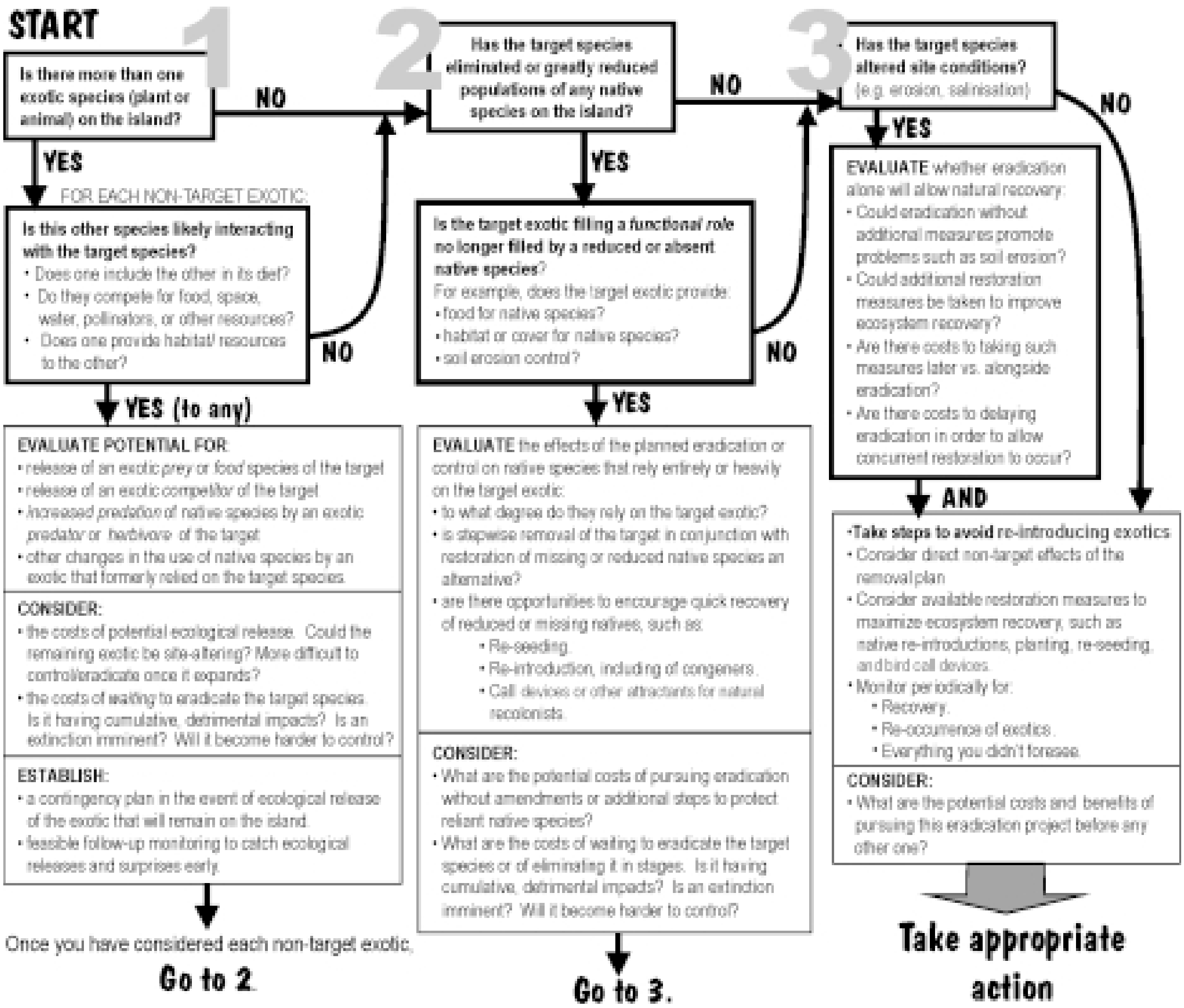

Fig. 3 A prototype planning guide for averting unexpected eradication outcomes on islands. To walk through issues to consider for a particular eradication project, start with the upper left-hand box (START). Use your yes/no answers to choose which arrow to follow from each *question box* (thick outline). When you reach a *guideline box* (thin outline), read through it then follow the arrow from it to the next step.

and pig eradication will almost certainly reduce widespread areas of bare soil throughout the island, stemming topsoil erosion and aiding recovery of heavily impacted native species such as a potentially endemic variety of *Opuntia englemanii* (Fig. 4; pers. obs.). However, small to significant (>1 ha) patches of up to seven new noxious exotic weeds, including Bermuda grass (*Cynodon dactylon*) and bufflegrass (*Cenchrus echinatus*), exist near the island's inhabited military garrison. If these exotic plants spread over large areas of the island when released from herbivory pressure, they may become impossible to ever remove. Little information exists, however, to suggest whether exotic plant release in this setting is a likely outcome. A preemptive, costly, multi-year weed eradication attempt before eradicating herbivores could safeguard against potential exotic plant spread but is riskier than immediate herbivore eradication from the standpoint of reversing declines in seabird populations, soil conditions, and certain native plant populations. The many unknowns complicating this weighing of options include how native and exotic plant species will respond to herbivore removal, whether the spread of new exotics would negatively affect island biodiversity and functioning more than feral herbivores do, and how imminent are threats of extirpation or extinction to certain native species.

Planners cannot, in this very typical kind of situation, presage the optimum path to complete island restoration. What they *can* do is to choose a first step wisely, identify outcomes to this step that they absolutely want to avoid, qualitatively evaluate the likelihood of such outcomes, and take steps to prevent them. Eradications have been a singularly effective conservation tool on islands; they have helped save numerous species from extinction and numerous ecosystems from collapse. Eradications can do more. As eradication advances in a technical sense, with ever-improving baits and traps, hunting strategies, and hard tools, its practitioners should also strive towards the state-of-the-art in an ecological sense. This means taking ad-

Fig. 4 Prickly pear (*Opuntia englemanii*), once widespread in the lowlands of Clarion Island, Mexico, now survives only on rocky outcrops that protect it from feral herbivores.

vantage of a different set of tools – monitoring, species reintroduction and translocation, revegetation and erosion control, and qualitative, systems-level ecology. It means placing more emphasis on achieving and verifying, not just identifying, long-term ecosystem restoration goals. As knowledge about the ecological context of eradications evolves alongside technical expertise, the conservation value of invasive species management can only grow.

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ABSTRACTS

The following pages contain abstracts for papers which were presented at the conference, but for which the authors have chosen not to prepare a full written paper.

These abstracts are given in the alphabetical order of the person who presented the paper at the conference (the name shown in bold type). The address of the presenter is given with each abstract.

Removing a diverse suite of invasive threats to recover an endangered Hawaiian bird species and its dry forest habitat

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Recovery of the Hawaiian forest bird community that includes the endangered palila (*Loxioides bailleui*) requires removing a wide array of invasive pests and weeds from subalpine dry forest habitat on Mauna Kea volcano. Palila are threatened by predators and food competitors, and their habitat is threatened by aliens that browse native vegetation, increase fire fuel levels, and suppress forest regeneration. Due to these and other factors, most palila are concentrated in only 30 km^2 of habitat. In addition, the palila is a seed specialist that obtains most of its food resources from mamane (*Sophora chrysophylla*), an endemic leguminous tree that is sensitive to browsing by feral sheep, mouflon sheep, and cattle. Episodically during the past two decades, sheep and mouflon populations have been reduced, resulting in mamane regeneration in many areas. However, annual counts suggest that the palila population may not benefit from these habitat improvements until saplings have grown larger. Invasive annual grasses suppress mamane regeneration and accumulate as fire fuel, and an alien vine overgrows trees. We are mapping the distribution of these and other weeds to facilitate control strategies. We are investigating the ecology of alien mammals to develop control priorities and strategies. Feral cats and black rats destroy many palila nests, and the tendency of birds to roost repeatedly in the same trees may increase their vulnerability to mammalian predation. Cats are readily trapped, but tracking studies indicate that immigrants will arrive from far outside control areas. Cats on Mauna Kea seem to prey more on birds than on house mice; therefore, reducing mouse populations may have little impact on cat numbers. Mice are abundant in palila habitat and are active in tree canopies, but their potential threat to palila or the forest is unclear. Threats to insect food resources of palila include alien wasps that parasitise and prey on caterpillars. Ants also are spreading into palila habitat and threaten the entire insect community. Protecting and enhancing the main palila population and re-establishing another population elsewhere on Mauna Kea depend on the effectiveness of reducing the impacts of this complex suite of invasive aliens.

Introduced Neotropical tree frogs in the Hawaiian Islands: Control technique development and population status

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Two species of Neotropical tree frog (*Eleutherodactylus coqui* and *E. planirostris*) have been introduced into the Hawaii Islands via the horticulture trade. Since 1997 frog colonies within the state have rapidly spread accidentally and intentionally and frog abundance within colonies has grown rapidly. Colonies of these frogs are currently known from 150+ locations on the island of Hawaii, 35+ on Maui, 5+ on Oahu, and one on Kauai. Although these frogs were originally restricted to horticulture sites, they are now found in residential areas, resorts and hotels, and public lands. Individual frogs or frog colonies have been verified at sites ranging from sea level to over 3500 ft. Within their native range, where their populations may be restrained via predation and other natural checks, they may reach densities of 20,000 frogs/ha and consume an estimated 140,000 prey items/night. Given the current population irruptions of these frogs in Hawaii, similar densities could be reached or exceeded. Given the high potential biomass of introduced frogs, there are realistic ecological and anthropogenic concerns associated with the spread of these frogs. Currently, there are limited techniques to control these animals. Research has been conducted to evaluate the efficacy of various mechanical and chemical techniques for frog control. Thus far, hand capture and trapping have proven labour intensive for frog control in sites with locally moderate to high frog densities. In collaboration with state pesticide regulatory and wildlife management agencies, we tested 30+ compounds (registered insecticides, surfactants, human pharmaceutical compounds, and food additives) in the laboratory to determine their efficacy for tree frog control. In these trials, caffeine and water solutions proved to be the only compounds that could effectively be used for tree frog control. Currently, field trials are being conducted to evaluate the efficacy of a direct spray application of a concentrated caffeine and water solution for tree frog control on 0.1 - 0.5 ha tree frog-infested plots. If these trials are successful, it is hoped that management agencies in the State of Hawaii will be able to reduce the spread and potential impact of these pest species on a landscape scale.

Tackling tussock moths: strategies, timelines and outcomes of two programmes for eradicating tussock moths from suburbs of Auckland, New Zealand

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Suburb-wide, aerial sprays of an organic insecticide (*Bacillus thuringiensis*) were applied to an infestation of the white-spotted tussock moth (*Orgyia thyellina*) in Auckland, New Zealand, followed by intensive monitoring for the remnant of the population. Caged females in sticky traps caught small numbers of wild males from a tightly localised area. This area was sprayed from the ground and with helicopters. Ground searches for eggs and caterpillars found nothing. The catch in the female-baited traps decreased as the summer programme of targeted sprays continued. A synthetic pheromone was identified from an international effort and deployed in a large number of sticky traps in the second year of the programme. No males were caught. The lures were subject to an independent quality assurance test for attractiveness. The moth was declared to have been eradicated and was not found again during the following two years. A second species of tussock moth (*Teia anartoides*) was found in two Auckland suburbs the following year. Localised ground spraying of infested areas with synthetic insecticides was followed by ground searches for caterpillars. These searches have yielded a steady number of caterpillars for a 12-month period. No attempt to use natural pheromone sources was made, and an attempt to identify the sex pheromone by a local group has failed. The moth continues to be present at the time of writing. The use of pheromones is concluded to be the key tool in any attempt to eradicate an invading moth species.

Recovery of invertebrate populations on Tiritiri Matangi Island, New Zealand following eradication of Pacific rats (*Rattus exulans*)

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The effects of kiore, or Pacific rat (*Rattus exulans*) on indigenous species has historically been based on anecdotal accounts and circumstantial comparisons. The eradication of kiore from 220ha Tiritiri Matangi Island in 1993 provided an opportunity to obtain empirical data on the effects of this rodent on invertebrates. Long term monitoring of ground invertebrates began three months before the removal of kiore and continued for five years following removal. Pitfall traps were set in a mature broadleaf forest remnant and in a younger regenerating forest. Larger numbers of invertebrates were caught in the mature forest and these also increased to a greater degree after rat removal. Capture rates of several large (>10 mm) species increased during the study, including ground weta (Orthoptera: Anostostomatidae) and several species of prowling spider (Araneae: Miturgidae). Capture rates of other species that changed over time appear to be correlated with weather that varied dramatically during the period of monitoring. Seasonal changes are reported and the life histories of large flightless, nocturnal ground-dwelling invertebrates are correlated with kiore eradication.

Restoration of tree weta (Orthoptera: Anostostomatidae) to a modified island

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Management of Korapuki Island in the Mercury Islands, Eastern North Island, New Zealand, has been centred around restoration activities following eradication of kiore, the Pacific rat (*Rattus exulans*) and rabbits (*Oryctolagus cuniculus*) during 1986-1987. As part of this restoration four lizard species have been transferred to the island and recently the first invertebrate, the Auckland tree weta (*Hemideina thoracica*), was transferred from Double Island, also within the Mercury Islands, to Korapuki Island. Further invertebrate transfers are planned. With relatively few invertebrate transfers recorded in New Zealand, the study aimed to develop protocols for the translocation of invertebrate species. It also aimed to use monitoring methods that measure impacts on the source population and success of the transfer. Artificial roost sites, in the form of wooden blocks with a single hole, were successfully used to monitor both the source population and the transferred population, in addition to facilitating the actual transfer. The criteria used to assess the potential release sites on Korapuki Island, details of the transfer, changes in populations on both islands, and life history information required to assess the likely risks are presented.

Control of cats on mountain "islands", Stewart Island, New Zealand

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The southern subspecies of the New Zealand dotterel (*Charadrius obscurus*) is currently restricted to nesting areas on the bleak alpine mountaintops of Stewart Island. By the early 1990s the species had declined to a total population of 65 individuals. The principal cause of the decline was attributed to predation by feral cats (*Felis catus*). A cat-control programme was initiated in 1992. The programme involved a perimeter of bait stations set up at the bush-line. Poison baits for cats were presented in these stations during spring and summer, when dotterels were nesting. Research suggests that cats are not resident year-round in the sub-alpine scrub. They generally stray into the sub-alpine scrub and above the bushline during the summer "low" in abundances of rats, their principal prey on Stewart Island. Little alternative prey is available during summer. The cat control appears to have been successful, as the population of dotterels has expanded to 170 individuals by April 2000. Research into the habitat preference of cats is continuing with a view to more efficient use of resources for ongoing cat control.

The status of invasive ant control in the conservation of island systems

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Introduced ants have often been responsible for significant ecological disruption in both continental and insular systems. In the oceanic islands of the Pacific, in particular, ants have long been implicated in the wholesale extirpation of the native lowland arthropod fauna. Three tramp species have been identified as especially invasive and detrimental to native ecosystems: the Argentine ant (*Linepithema humile*), the big-headed ant (*Pheidole megacephala*), and the little fire ant (*Wasmannia auropunctata*). Others are undoubtedly important as well, and on islands where these three species do not occur, the ecological effects of other community-dominant species needs to be assessed. While removal of invasive ant species would benefit many natural areas, attempts at eradication have typically been unsuccessful. The history of control efforts has most often involved the use of broad-spectrum pesticides in agriculture and urban settings, with active ingredients shifting as regulatory standards have changed. More recent conservation efforts in the Galapagos and Hawaii have employed the toxicant hydramethylnon in attempts to eradicate the little fire ant and the Argentine ant from natural areas. Results indicate that at least one effort in the Galapagos may have been successful against the little fire ant, but on experimental plots in Hawaii eradication of the Argentine ant has failed. The present focus in Hawaii has therefore shifted to strategies for suppressing further invasion, and this has met with moderate success. We report these results here. Alternative approaches, including the disruptive use of juvenile growth hormones and semi-chemical pheromones, have been limited. Implemented alone, these techniques generally have not shown promise of achieving eradication. This leaves the state of invasive ant control unresolved, especially because the current techniques available are appropriate only in certain situations. As a new campaign is being undertaken against the Argentine ant in New Zealand, and as problematic tramp species in general continue to expand their ranges, the need for renewed investigations into ant eradication techniques is critical. We discuss briefly some novel integrative approaches that might be attempted.

The effectiveness of weeded and fenced 'Conservation Management Areas' as a means of maintaining the threatened biodiversity of mainland Mauritius

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Mauritian native ecosystems continue to be degraded by the action of alien plants and animals. Lack of management is not an option if Mauritius' unique biodiversity is to be maintained. Weeding of alien plants and the fencing out of deer and pigs was first recommended for conservation in the 1930s. Ten weeded and fenced 'Conservation Management Areas' (CMAs) have been set up in a variety of ecosystem types. Predator control is also practised in some CMAs. The effects of management on native flora and fauna have been quantified in several upland CMAs. Consistent weeding and maintenance of fences appears to result in a spectacular regeneration of native flora. In the Brise Fer 'Old Plot', first weeded and fenced in 1987, a minimum of between 53% and 68% of native tree taxa are regenerating. Regeneration of

some taxa was probably prevented by mammals that cannot be excluded by conventional fences. The diversity of native seedlings and saplings is relatively low in a more recently managed part of Brise Fer and in the nearby Mare Longue CMA respectively. In the former this may be due to the fact that several deer were inadvertently fenced into the CMA for over two years. In the latter, rocks were not placed at the foot of the fence, thus allowing pigs to burrow into the plot. Native butterflies were on average 19 times more abundant in the surveyed CMAs than in non-managed areas while results for native birds were equivocal. In contrast, densities of some native snail groups were lower in CMAs. This may be due to the effect of persistent rat poisoning and the change in habitat after initial weeding. Current CMAs can be highly effective if the fencing is of a consistently suitable standard, and if any incursions of deer and pig are dealt with rapidly. Weeding methods may have to be modified to minimise non-target damage. Non-regenerating or negatively impacted species may have to be managed individually. An alternative or complement to this would be the use of predator-proof fences.

Preparation for the eradication of Norway rats (*Rattus norvegicus*) from Campbell Island, New Zealand

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Norway rats (*Rattus norvegicus*) have populated Campbell Island in the New Zealand subantarctic for nearly 200 years. During this time they, in combination with feral cats which have since died out, have had a devastating effect on the island's fauna, marooning several species of bird to the small rat-free islands around the coast and probably causing the extinction of several other undiscovered species. At 11,300 ha the attempt to eradicate rats from Campbell Island will be the largest ever undertaken. The island's size coupled with its location in the furious fifties (700 km south of the New Zealand mainland) renowned for their strong winds and frequent rainstorms, means the attempt will be stretching the boundaries of current technology. In order to make the eradication logistically feasible, the margin for error that has been build into all previous eradications has had to be significantly reduced. Instead of two bait drops totalling 12kg/ha as is usually used, Campbell will be done with a single drop, but with a 50% overlap to eliminate the risk of gaps, totalling 6 kg/ha. This technique was tested in 1999 with a 600ha field trial carried out on the island. Rhodamine dye showed that all the rats in the baited area ate bait and would therefore have been killed. While there is only 70 hours of bait dropping required and three helicopters will be used, the short daylight hours on Campbell during the winter and the predictably bad weather, mean that the project team must plan to be on the island for up to three months. Non-target issues are minimal, with the only priority species at direct risk being southern skuas (*Catharacta skua*), which fortunately are absent from the island until mid August. However if the drop is delayed not only will skuas be affected but so will the large colonies of mollymawks which nest at the north end of the island and present a real risk to the helicopters. The drop will be carried out in July - September 2001 with no follow-up until 2003. There will only be one attempt and we either succeed or fail.

Island quarantine – prevention is better than cure

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While there is currently an international focus on the eradication of introduced invasives (particularly rodents), it is easy to forget that usually the easiest, cheapest and often the only way to avoid the significant impacts that invasives can have on an island ecosystem, is to prevent them getting to the island in the first place. There are numerous examples of "near misses", where rodents in particular have made it onto islands or have only failed to do so by sheer luck. That these incidents did not result in the establishment of the predators can only be put down to good fortune. It is vital that in future, precautions are put into place that mean that we don't rely on luck to keep islands free of introduced species. To date, quarantine precautions have focussed largely on rodents, and they have been the flagship of the quarantine battle. This has been very productive as everyone hates rats and can understand/relate to the damage they can do on an island. However, with increased awareness and knowledge, it is apparent that other species can pose a greater, albeit not so obvious, risk. These include invertebrates, plants and even microorganisms. Standard precautions such as sealed containers work well for rodents but do little to prevent the introduction of a vast and ever increasing number of weed species and invertebrates. This risk is at a new level and relies heavily on the individual taking responsibility. Basic precautions such as scrubbing footwear and checking pockets are simple and can significantly reduce the risk, but are often overlooked. All visitors to islands (tourists, researchers, and managers) pose a quarantine risk, and a quarantine plan must be practical for all the situations relevant to the specific islands. Currently there are significant resources being dedicated to trying to stop invasives, rodents in particular, getting established once they make it to an island. However, these contingencies are often expensive and are at best unreliable. Further research is required on the effectiveness of the various options (e.g. traps vs toxin), and also on the behaviour of rats reaching a new island.

The role of parasitoids in eradication or area-wide control of tephritid fruit flies in the Hawaiian Islands

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Koinobiont larval endoparasitoids (Hymenoptera:Bracondae:Opiinae) of tephritid fruit flies have been used with variable success rates in classical biological control, but have not previously been considered as useful adjuncts to eradication programmes. Successful eradication of tephritid flies from other Pacific Islands has been accomplished by utilisation of semiochemical-toxicant combinations (male annihilation) and/or the sterile insect technique (SIT). However, pilot projects designed to show feasibility of medfly (*Ceratitis capitata*) eradication on the Hawaiian island of Kauai failed because of the inability to achieve adequate overflooding ratios of sterile:wild males. Medfly population reduction sufficient to achieve suitable overflooding ratios is not possible with insecticides due to the location of populations in remote and environmentally sensitive areas. Population dynamics models demonstrate the synergistic effect of combined augmentative parasitoid releases with SIT. The release of new species or strains of parasitoids that are self-perpetuating and dispersing would be more cost-effective. The potential is demonstrated by *Fopius arisanus* from Asia, which causes over 95% egg mortality of Oriental fruit fly (*Bactrocera dorsalis*) in guava. Comparable levels of parasitism for melon fly (*B. cucurbitae*) in cucurbits might make eradication in Hawaii feasible.

Response of forest birds to rat eradication on Kapiti Island, New Zealand

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Five-minute bird counts were used to determine whether the eradication of Pacific rats or kiore (*Rattus exulans*) and Norway rats (*R. norvegicus*) from Kapiti Island in 1996 had any measurable impact on the diurnal forest bird community. Counts undertaken quarterly from April 1999 to January 2001 were compared with counts undertaken using the same methodology from April 1991 to January 1994. At least four species appear to have increased since rat eradication: red-crowned parakeet (*Cyanoramphus novaezelandiae*), robin (*Petroica australis*), saddleback (*Philesturnus carunculatus*), and bellbird (*Anthornis melanura*). None of the 15 species investigated showed evidence of a consistent decline since rat eradication, although two (tui (*Prosthemadera novaeseelandiae*) and tomtit (*Petroica macrocephala*)) were less conspicuous than in 1991-1994 in four of the eight count sessions completed to date. Weka (*Gallirallus australis*) were adversely affected by the rat poisoning operation, but had recovered to pre-eradication levels by 1999. The present series of counts will be completed in January 2002.

Sustained control of feral goats in Egmont National Park, New Zealand

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Egmont National Park (33 540 ha) is a forested mountain ‘island’ surrounded by a ‘sea’ of farmland. Feral goats have been present in the Park since *c.* 1910. Control efforts have been ongoing since 1925, making it one of the longest-running sustained vertebrate pest control operations in the world. Although helicopter-based hunting has proven effective at reducing goats above timberline, most of the Park is forested and the primary method of control in this habitat has been ground-based hunting with dogs. We used indices of hunting effort (days hunted) and goat population density (goats killed/days hunted), to investigate trends in the goat population in response to management during the period 1961-1999. Annual hunting effort generally increased over the period 1961-1986 but, following a change in the management organisation in 1987, has since declined. Goat density was highest in the earliest years of control (*c.* seven goats killed/day) and steadily declined until 1987 (0.8 goats killed/day). Post-1987 the population has been maintained at low densities (<2 goats killed/day). The likely consequences of alternative strategies for allocating hunting effort on goat densities will be discussed.

Pacific rats: their impacts on two small seabird species in the Hen and Chickens Islands, New Zealand

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The Hen and Chicken Islands support introduced Pacific rat (*Rattus exulans*) populations and also remnant populations of two small burrow-nesting seabirds, the summer breeding Pycroft's petrel (*Pterodroma pycrofti*) and the winter-nesting little shearwater (*Puffinus assimilis haurakiensis*). The sequential eradication of kiore from the larger Chickens in the 1990s provided an opportunity to measure the responses of these seabirds to kiore removal. The following hypotheses were tested: (1) Breeding success of the two seabird species is not limited by kiore presence, (2) breeding success is not limited by the presence of tuatara (*Sphenodon punctatus*) an endemic predatory reptile, (3) the two seabird species are not in competition with each other. Two study islands were used: Coppermine and Lady Alice Island. Study burrows were checked early and late in the seasons to determine breeding success. Success was significantly lower when the study islands contained kiore: little shearwaters averaged a 16% breeding success in the presence of kiore and 61% in the absence of kiore, while Pycroft's petrels averaged a 33% breeding success in the presence of kiore and 57% in the absence of kiore. Contemporaneous data for the two islands enabled other factors such as food supply and heavy rainfall to be eliminated as confounding variables. For example, the lowest breeding successes of little shearwater (5%) occurred in a kiore-present scenario for two years on Coppermine, but in the same two years productivity was high on kiore-free Lady Alice Island. Similarly, for Pycroft's petrels, the lowest years of breeding success were in kiore-present scenarios, but in the same years there was significantly higher productivity in the kiore-free scenarios. The presence of tuatara in burrows did not significantly influence the breeding success of these seabirds, at least in a post-kiore scenario. However, in burrows used by both species of seabirds, late-fledging little shearwaters disrupted the nesting of Pycroft's petrels, causing some pairs to be displaced to other burrows or abandon nesting for the season. Currently, the effects of this competition on Pycroft's petrel are small and more than compensated by their increased productivity following kiore removal. In conclusion, these findings demonstrate clear negative impacts of kiore on small seabird productivity. They are consistent with the recorded decline in seabird populations on the Hen and Chickens Islands during the 20th century. Count data from Lady Alice Island from 1992 to 2000 indicate that this population decline has been halted and apparently reversed.

Seabird re-colonisation after cat eradication on equatorial Jarvis, Howland, and Baker Islands, USA, Central Pacific

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In 1990, the last feral cat (*Felis catus*) was removed from Jarvis Island National Wildlife Refuge in the Central Pacific Ocean. Cats were removed from two other equatorial islands: Baker Island in the 1960s and Howland Island in 1986. Introduced during the 1930s, cats had extirpated some species of terns, small procellarids, Pacific (*Rattus exulans*) and Norway (*R. norvegicus*) rats from Jarvis, Howland and Baker Islands. After cat and rat eradication, previously extirpated seabirds (blue-gray noddies (*Procelsterna cerulea*), Christmas and Audubon's shearwaters (*Puffinus nativitatis* and *P. lherminieri*) and white-throated storm-petrels (*Nesofregetta fuliginosa*)), have re-colonised Jarvis Island. Baker Island has been re-colonised by wedge-tailed shearwaters (*P. pacificus*) and hundreds of thousands of birds that moved from Howland Island. Small tern populations are returning to Howland Island as well as increased numbers of wintering shorebirds. Even though the predation-free period has been longer for Howland and Baker than Jarvis Island, fewer new species have re-colonised them, probably because of the great distance to other colonies that could serve as a source.

Direct and indirect effects of house mice on declining populations of a small seabird, the ashy storm-petrel (*Oceanodroma homochroa*), on Southeast Farallon Island, California, USA

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There is concern over severe population decline of the ashy storm-petrel (*Oceanodroma homochroa*) on Southeast Farallon Island (SEFI), California (37°N 123°W). Evidence from a mark-recapture analysis suggests that a primary cause of this decline is increased predation on this species, whose main predators include expanding populations of western gulls (*Larus occidentalis*) and migrant burrowing owls (*Athene cunicularia*). There is evidence that the introduced house mouse (*Mus musculus*) may occasionally prey upon ashy storm-petrel eggs and chicks, although the extent of this is unknown. Owl arrival in the fall coincides with the peak mouse population, but with decreasing food supplies in the late winter, the mouse population reaches a low point. When this occurs, the wintering owls lose a primary food source and may shift their diet from house mice to ashy storm-petrels, which arrive to SEFI in early spring to begin their breeding cycle. Thus, the indirect effect of mouse presence on ashy storm-petrel populations, through burrowing owls, is perhaps more severe than the direct effects. The U.S. Fish and Wildlife Service is currently considering a proposal to remove house mice from the island. Justification of this action will rest heavily on documentation of adverse affects of mouse presence on the natural ecology of SEFI. Before eradication plans are implemented, all factors, both direct and indirect, must be considered.

Managing pest mammals at near-zero densities at sites on the New Zealand mainland

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Six "Mainland Island" projects are being undertaken by the Department of Conservation at sites on the North and South Islands of New Zealand. A feature of these projects is the range of pests targeted for control and the relatively low pest densities achieved as a result of pest control operations. These early results are significant in that they suggest that effectively managing the impacts of mammalian carnivores and herbivores is achievable on the mainland as well as on offshore islands. The challenge now is to develop more efficient pest control regimes so that conservation outcomes may be sustained. Better targeting and timing of pest control and more effectively controlling pest re-invasion rates will result in further advances in our capacity to conserve native biodiversity. In view of the urgency in implementing more effective conservation programmes to arrest further declines, and recognising our inability to predict ecological outcomes from intensive, multi-pest control programmes, an adaptive experimental management approach has been proposed to enhance our understanding of pest impacts; ecological responses are enhanced as part of the management process.

Control of feral goats (*Capra hircus*) on Santa Catalina Island, California, USA

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Santa Catalina Island, a mountainous, 194 km^2 island is the third largest of the eight California Channel islands. In addition to numerous endemic species, it also has a resident human population of approximately 4000 people and nearly 1,000,000 visitors per year. The Santa Catalina Island Conservancy owns and manages 88% of the island with a primary goal of natural resource protection while still allowing appropriate public access. Among Catalina's non-native mammal species are feral goats (*Capra hircus*) which were well established by the mid 1800s and may have reached a population high of 30,000 in the 1930s. Impacts by goats on natural resources have been severe, including destruction of endemic plant species and island plant communities, increased erosion, and soil compaction. Although sport hunters and island resource managers removed large numbers of goats throughout the years, sizeable populations remained until the 1990s. From 1990 to 1994, ground and aerial hunting removed over 7700 goats from the island, but due to lack of funding the programme was stopped after approximately 95% of the goats were removed. In 1996, a new effort was initiated to remove all goats from the west end of the island. By early 1998, the only goats known to be in this area had telemetry tracking collars attached and the programme was expanded to include the rest of the island. During the next

six months over 600 goats were removed by hunting. Following a community forum in January 1999, an outside animal welfare organisation submitted a live capture proposal. The Conservancy Board elected to suspend hunting to try the proposal. In the fall of 1999, 121 goats were captured and shipped off the island. In January 2000, permission to resume hunting was granted and 66 goats were removed. Shortly thereafter, another live capture proposal was submitted and the Board elected to follow a live capture programme for all remaining goats on the island (estimated 25-30). By the start of 2001, all uncollared goats should be removed and we hope to remove the collared goats by the end of 2001, thus reaching the objective of zero goats.

Control of the invasive exotic yellow crazy ant (*Anoplolepis gracilipes*) on Christmas Island, Indian Ocean

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The exotic invasive ant *Anoplolepis gracilipes* was accidentally introduced to Christmas Island between 1915 and 1934. It remained in relatively low numbers until 1988 when an isolated infestation of very high densities of ants was discovered. In 1998 several more infestations were found. Infestations range in size from less than 1 ha to over 100 ha. Currently about 1400 ha or 14% of the forest is infested. In areas of infestation *A. gracilipes* forms extensive multi-queened supercolonies where high densities of workers are sustained on the forest floor and on most plant surfaces, including rainforest canopy species. These infestations have serious impacts on the integrity of the rainforest ecosystem of Christmas Island by eliminating the dominant red crab (*Gecarcoidea natalis*) from infested areas. The red crabs control seedling recruitment and litter breakdown, and their removal results in a rapid transformation of the rainforest ecosystem in terms of habitat structure, species composition, and ecosystem processes. The ant infestations also pose a serious threat to endangered species of birds and reptiles. Parks Australia has developed a methodology for a chemical control programme. The key requirements of this programme are that the bait be: (a) highly attractive to ants such that they monopolise the baits, (b) slow acting so that maximum transfer of bait occurs among individuals, (c) effective over a wide range of concentrations, and (d) not detrimental to non-target species. A number of chemicals and attractants were tested and the most effective bait was fipronil in a fish protein base. Widespread distribution at a rate of 0.5 grams active ingredient per hectare on half hectare plots demonstrated that this bait was effective in reducing ant densities, wiped out ant nests, and had no detectable non-target impacts. Larger-scale baiting is currently underway. While complete eradication of *A. gracilipes* from Christmas Island is probably an impossible task, initial baiting trials have shown that it may be possible to reduce ant densities to levels where red crabs and ants can coexist. A concurrent research programme is being undertaken along with the control program in order to provide better information for the management of this issue.

Preventing rat introductions to the Pribilof Islands, Alaska, USA

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The Pribilof Islands have about three million nesting seabirds, a million northern fur seals, an endemic shrew, and other wildlife. Rat introduction would greatly reduce bird and shrew populations and might transfer diseases to humans and wildlife. The islands have been inhabited since 1786 and, although the lack of harbours impeded rodent introduction, house mice became established on St. Paul in 1872. In the early 1990s harbours were constructed on both St. George and St. Paul Islands. A boom of commercial fisheries soon followed and eventual rat introductions seemed a certainty. With the objective of keeping the Pribilofs rat free, a prevention programme was begun in 1993 based on cooperation with local communities, government agencies, and industry. The programme consists of maintaining trap and poison stations, community education, local shipwreck response capabilities, outreach to vessels to make them rat free, and regulations. Over 450,000 trap nights have passed and several rats have been killed on the St. Paul docks, but there is no evidence of rats becoming established anywhere in the Pribilof Islands. Improved design of preventive stations has decreased maintenance needs. Snap traps have been more effective than poisons, but have caused minor loss to non-target species. Both techniques are recommended. The local communities are taking increasing ownership in the programme and it appears fewer ships using the Pribilofs carry rats. Unless there is a major advancement in rodent removal technology, the prevention programme will have to be maintained forever. It is too early to be certain that the program is adequate to protect the Pribilof Islands, but as each rat-free year passes, hopes are rising.

Ecological restoration of islands in Breaksea Sound, Fiordland, New Zealand

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New Zealand has long been renowned internationally for its innovative conservation strategies. Translocation as a conservation management technique was pioneered in Fiordland between 1894 and 1900 when Richard Henry undertook 700 transfers of a range of vulnerable birds such as kakapo (*Strigops habroptilus*) and kiwi (*Apteryx australis*) to Resolution Island (and smaller adjacent islands) from nearby mainland sites in Breaksea and Dusky Sounds. This far-sighted project was abandoned when introduced stoats reached the area. Biological surveys of islands in Breaksea Sound in the 1970s resulted in an ambitious island restoration project in which Norway rats (*Rattus norvegicus*) were eradicated from bush-clad Hawea Island (9 ha) in 1986 and rugged Breaksea Island (170 ha) in 1988. Before poisoning on Breaksea Island, South Island robins (*Petroica australis*) were transferred to Hawea Island as a precautionary measure. The resultant population is the densest recorded, and they have even dispersed across 300 m of open water to neighbouring Wairaki Island (3 ha). South Island saddlebacks (*Philesturnus carunculatus*) were released onto Breaksea Island in 1992 with similar success and yellowheads (*Mohoua ochrocephala*) are confirmed breeding following an experimental transfer from the mainland in 1995. We undertook some of the first experimental translocations of lizards and invertebrates: Fiordland skinks (*Oligosoma acrinasum*) being released onto Hawea Island in 1988, and knobbled weevils (*Hadramphus stilbocarpae*) and flax weevils (*Anagotus fairburnii*) transferred to Breaksea Island in 1991. A programme to monitor ecological change, using several key, indicator species of flora and fauna, was set up before poisoning to document the benefits of eradicating rodent pests. Natural dispersal of Fiordland skinks onto Breaksea Island from a nearby rock stack was confirmed within two years, and recovery and/or natural dispersal of vulnerable flightless mega-weevils on islands in Breaksea Sound provide further examples of the dynamic ecomarine interface. Despite a lack of funding, monitoring has been maintained, albeit at a reduced level, by using eco-tourism to provide the essential but expensive logistic support and field assistance needed to undertake research in such a remote location.