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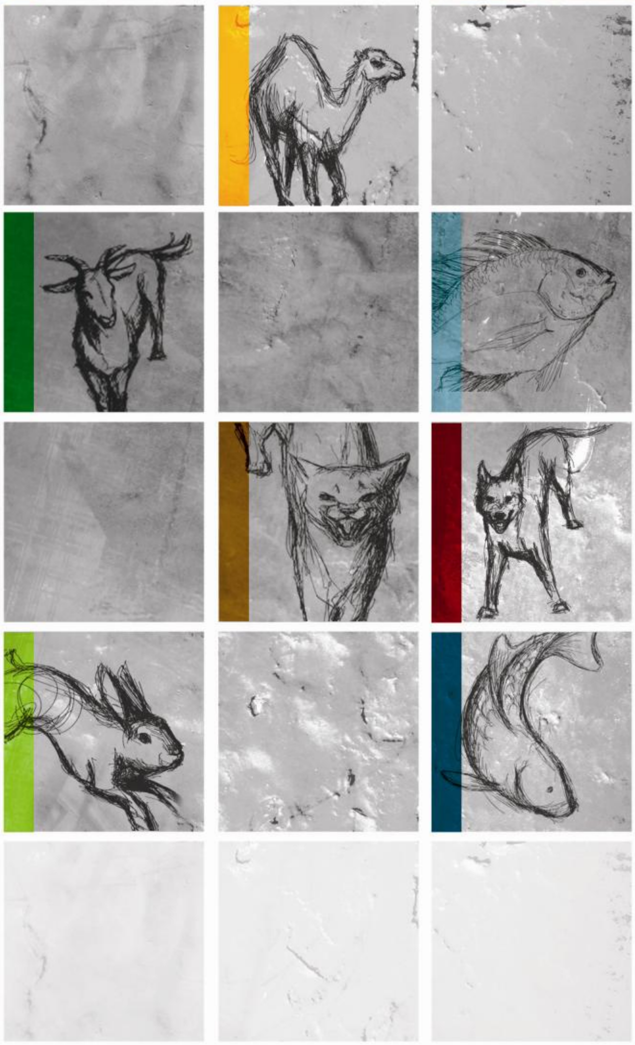

Eradications of vertebrate pests in Australia

A review and guidelines for future best practice

Stephen Gregory
Wendy Henderson
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2014

Invasive Animals CRC
THE UNIVERSITY of ADELAIDE
An Australian Government Initiative
CRC AUSTRALIA

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Front cover photo: Nick Markopoulos and Brenton Florance from the Kangaroo Island Natural Resources Management Board (South Australia) during the Kangaroo Island feral goat eradication program (2006 - ongoing); Photo source: Pip Masters.

Foreword

Most of the public did not recognize the phenomenon of biological invasion as one of the major ongoing global changes until the late 1980s, although Australia was in the vanguard because of obvious impacts of such invaders as the European rabbit and South American prickly pear, introduced in the late 18th century, the camel, brought in the 19th century, and the cane toad from Latin America, which arrived in the early 20th century. Until recently, for the most part, both the public and scientists recognized damaging impacts of a few particular invaders and attempted to deal with them on a case-by-case basis with varying degrees of success. Many attempts at managing invasions were not recorded or at best appeared in the gray literature; therefore, until recently not many lessons were learned from both successes and failures that could be applied to later efforts. As the full depth and scope of invasions and associated impacts exploded in the public and scientific consciousness, this piecemeal approach to studying and managing invasions began to change, and surveys of various classes of invasions began to appear, along with books and journals wholly devoted to invasion processes, impacts, and management.

Management of invasions has largely fallen into three categories: (1) keep them out; (2) if they get in, find them (early, if possible) and attempt to eradicate or at least to contain them; (3) if they become widespread and numerous, attempt by various means (pesticides, herbicides, biological control, mechanical or physical removal) to maintain their populations at low densities. Australia, along with New Zealand, has been one of the leaders in both effort and technology to keep invaders out, spurred by early disastrous deliberate introductions such as the four cited above. However, even the best interdiction efforts will fail at times, either because stowaways and hitchhikers arrive or because a permitted introduction ends up causing unforeseen trouble. Further, the residue of many older introductions remains – rabbits and cane toads still abound in parts of Australia. Thus, the next possible line of defense – eradication – comes into play.

Eradication of invasive species – the complete removal of all individuals of a discrete population – has a curious history in invasion management. Policymakers and politicians love eradication, because it is an aggressive response to a perceived problem and because, if it works, even an expensive eradication campaign is cheaper than management costs and losses associated with the invader in perpetuity. Successful eradication stops the problem once and for all. Scientists and most managers, however, were until recently largely skeptical of eradication, on two main grounds. They thought the likelihood of success was low, except for invasions restricted to very small areas, and they felt that the collateral damage caused by eradication efforts was likely to be drastic, especially in comparison to the approach seen as a “green” alternative, biological control. This feeling was heightened by the fact that several eradication attempts were famous fiascos, failing to eliminate the target pest while devastating non-target species.

Yet, along with the high-profile failures were several less well-publicized successes. A recent tally shows that, of 344 attempts to eradicate populations of Norway rats, ship rats, and Pacific rats, 318 have been successful. So far, there have been at least 1,300 attempts to eradicate vertebrate species on islands worldwide; Australia and New Zealand far outstrip other nations in these efforts, with over 300 island campaigns each. The first successful vertebrate eradication appears to have been Australia’s removal of goats from Norfolk Island in 1857. Australia’s primacy in this approach is perhaps not surprising, as Australia has over 8,000 islands and a long history of notable invasions.

A striking recent global development in eradication of invasive vertebrates, invertebrates, plants, and pathogens is attempts on larger and larger islands, and even on continents.

Probably the most remarkable success is the global eradication of rinderpest, a devastating viral pathogen of ungulates, in 2011. Small islands of course present the advantages of relatively small target population sizes and, if they are sufficiently isolated, they are not likely to be reinvaded. Isolated and clearly delimited continental populations present some of the same advantages – this is why invasive fishes from lakes are frequently targeted. Of the nearly 300 mainland vertebrate populations targeted for eradication in Australia, the great majority were restricted to small, well-defined areas, in some instances exclosures fenced to prevent reinvasion. However, as experience is gained with existing technologies, and new technologies (e.g., autocidal genetic methods such as daughterless carp) are developed, there is every reason to think that eradication of larger and more widespread mainland populations will become feasible. Certainly recent successful eradications of vertebrates from very large islands (e.g., goats from 50,000 ha Santiago in the Galapagos) would have been viewed as pipe dreams as recently as a decade ago. And, as ever more invasive species are targeted on ever larger islands and mainland regions, there is every reason to believe that Australia will continue to be in the forefront.

Beginning in the 1990s, a number of authors, examining eradication attempts for various taxa in various regions, have attempted to determine which factors separate the successes from the failures. Most of these studies have resulted in verbal descriptions of keys to success – such factors as adequate resources to see a project through to completion, knowing that the last few individuals often require much more time and effort than those that have gone before. Or the ability to compel cooperation from all landowners and managers, because eradication is an all-or-none processes, and a single party refusing to allow bait or traps to be set can defeat an entire project. Such lists of keys to success invariably present particular cases that strikingly rationalize the list of claimed prerequisites, but without a comprehensive statistical analysis of all the data, it is difficult to assess the validity of the lists. This approach of eyeballing examples to extract key factors also makes it difficult to compare different resulting lists. It is perhaps not surprising that certain factors, like size of target area, show up on most lists, but that other factors differ. Within the last two decades, an increasing number of authors have conducted various statistical analyses to remedy these shortcomings.

Eradications of vertebrate pests in Australia is the most thoroughgoing effort so far to parse a substantial data set by rigorous statistical methods, including both univariate and multivariate analyses, and will surely inspire similar analyses of other eradication data sets, though none can probably approach the sample sizes for Australia. By combing the literature and unpublished reports, the authors have also greatly expanded the list of attempted eradications. For example, they have more than doubled the number of reported eradication attempts on islands compared to a recent published estimate, to 354 attempts on 188 islands. Combined with the detailed analyses of numerous individual cases, the statistical analysis of such a large data set allows the suggestion of an eradication “best practice” for vertebrates that constitutes a significant advance in this approach to invasion management.

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Summary

This report summarises the research conducted for the Invasive Animals Cooperative Research Centre project 'Success of eradication programs for vertebrate pest species on islands around Australia'. The project aimed to:

(i) identify factors that are reported to influence the success of vertebrate pest eradications;
(ii) collate data associated with the eradication attempts of vertebrate pests on Australian offshore islands and the mainland;
(iii) construct statistical models to test explanatory variables associated with the success (and failure) of eradication attempts of vertebrate pests on Australian offshore islands;
(iv) provide qualitative insight into mainland-eradication success (and failure) and compare this information with the lessons learned from island eradications; and
(v) review knowledge gaps (in both information and data) from past eradication attempts that can help improve the probability of future eradication success for researchers and wildlife managers.

Eradication is the complete and permanent removal of a 'pest' species from a defined area, and within a defined time period. Eradication data was collected for 650 vertebrate pest eradication attempts; 354 from 188 offshore islands and 296 from varions locations on the Australian mainland. The majority of vertebrate eradication attempts were reported to be successful (92.5% on islands). Overall, we found that increasing island ruggedness (a measure of topographic complexity) was the most consistently influential variable explaining differences in eradication success. Other variables of island eradication success, with less support in the top-ranked models, included: (i) increasing island size; (ii) greater elevational range; (iii) longer time that a pest population was present on an island; and (iv) closer distance from a source population.

In order to inform the future analysis and collation of data on eradication attempts we present a list (Appendix A) of the putative factors associated with the key aspects of vertebrate pest eradication attempts: (1) physical and biological factors at the location of the eradication attempt; (2) details of the vertebrate pest population eradicated; and (3) details of the eradication itself, i.e., methods and cost. To date, the majority of these fields have been extremely poorly recorded. A particular challenge for future analyses will be to record measures (and breakdowns) of the cost and consistency of eradication program efforts.

It is generally believed that the scientific and public pessimism surrounding the control of biological invasions is largely due to widely publicised management failures, especially failed eradications. We present a set of core "best practice" considerations for enhancing the success of eradication efforts, which can lead to increased public perception, as well as increased community and government support: (1) feasibility planning; (2) methodological and technological advances; (3) non-target species; (4) surprise effects; (5) pre- and post-eradication monitoring; (6) welfare and ethical awareness; and (7) documentation.

We are particularly grateful to the large number of people (see Acknowledgements) who extensively reviewed the report prior to publication, and who provided additional papers, unpublished data, and comments from the general to the specific.

1. Introduction

1.1 Motivation and objectives

Invasive species are among the greatest threats to global biodiversity (Vitousek et al. 1997), and constitute an unprecedented form of global change (Ricciardi 2007; Richardson and Ricciardi 2013). Invasive species are the cause of significant local species richness declines worldwide (Murphy and Romanuk 2014), and cause an estimated USD$1.4 trillion in annual economic losses (Pimentel et al. 2007). With ever-increasing globalisation and international trade, it is probable that these threats will continue to increase (Levine and D'Antonio 2003). Moreover, it is likely that human-induced global environmental change (including climate change and habitat alteration) will compound the problem by facilitating the establishment and spread of new invasive species introduced to formerly unsuitable regions outside their native range (Brook 2008).

One of the reasons that invasive species are such a vast problem is that once they have been successfully introduced, established, and spread, it is remarkably difficult to reverse the situation (Simberloff 2009). Besides doing nothing, management options include either controlling or eradicating the species. Of these alternative actions, eradication is preferred ultimately, because it is a solution with an end-point, and the harmful costs to the environment and economy are halted rather than minimised. Nevertheless, in the immediate-term, eradication can be more expensive and more harmful to native biodiversity, than control (Bomford and O'Brien 1995), and at large scales may not be currently feasible.

With a view to long-term biodiversity conservation, the objectives of this report are to: (i) identify factors that are reported to influence the success of vertebrate pest eradications; (ii) collate data associated with the eradication attempts of vertebrate pests on Australian offshore islands and the mainland; (iii) construct statistical models to test explanatory variables associated with the success (and failure) of eradication attempts of vertebrate pests on Australian offshore islands; (iv) provide qualitative insight into mainland-eradication success (and failure) and compare this information with the lessons learned from island eradications; and (v) review the state-of-knowledge (as well as gaps in both information and data) from past eradication attempts that can help improve the probability of future eradication success for researchers and wildlife managers.

By addressing these objectives we hope to equip policy and decision-makers (including government, industry, and the community) with the information necessary to assist with prioritising island and mainland regions where eradications of vertebrate pests have the highest chance of succeeding. In this respect, the report directly addresses Goal 3 Objective 3.2 and Key principle 9 of the *Australian Pest Animal Strategy* (Natural Resource Managment Ministerial Council Australia 2007):

Goal 3 *Manage the impacts of established pest animals*

Objective 3.2 *To identify and manage the impacts of pest animals on key assets.*

Key principle 9 *Management should be strategic in terms of determining where management should occur, timing of management, being proactive and using appropriate techniques.*

In this report, we are concerned with vertebrate pest species (mammals, birds, reptiles, amphibians and fishes), and particularly those whose populations are exotic in Australia and are not managed for commercial enterprise (i.e., neither native pests nor domestic populations of livestock species). We further discuss this distinction below.

1.2 Vertebrate pest species

We define a *pest* species as one that is having (or has had) a measurable negative impact on the host system where it occurs (www.feral.org.au/pest-species/) whether that impact is ecological, economic, or social. Pest species generally emerge in response to a perturbation within the system. One of the most common perturbations is the introduction of the pest species itself to a receiving environment outside its past (and present) geographical range. By definition, pest species are usually invasive and can spread, reaching large abundances, in their host environment (Figure 1).

Pest species that are not native to the host environment are commonly referred to as *exotic, invasive,* or *alien* species. Feral pests are a particular subset of these species. They are animals that were once livestock or domesticated species but have since established self-sustaining populations in the wild. Exotic pest species are usually introduced with the aid of human agency, and the reasons for their introduction are varied. In some cases, the species are reared for agriculture, including livestock species in Australia such as sheep *Ovis aries* and goats *Capra hircus*. Others were introduced because they aid humans in their work, such as horses *Equus ferus* or the drought-tolerant dromedary camel *Camelus dromedaries*, originally used to move and trade goods in arid environments. Some exotic species are introduced for nostalgic reasons, or to make the receiving environment seem more familiar, as was the case for many bird species introduced to Australia by the early European acclimatisation societies; including charismatic garden species such as the European Blackbird *Turdus merula* and Goldfinch *Carduelis carduelis*. Exotic species are still imported (and traded) as pets. In other cases birds and mammals were introduced for the pastime of hunting; such as the many wildfowl and game species – including, for example, mallard *Anas platyrhynchos*, European rabbit *Oryctolagus cuniculus*, European red fox *Vulpes vulpes*, fallow deer *Dama dama*, and rainbow trout *Oncorhynchus mykiss*. In rare cases, vertebrates have been introduced for biocontrol, including the cane toad *Rhinella marina* (formerly *Bufo marinus*) (Urban et al. 2008) and mosquitofish *Gambusia afinis* (Lloyd and Tomasov 1985). Finally, commensal species have taken the opportunity to hitch-hike as unwanted stowaways, particularly rodent and bird species such as black rats *Rattus rattus* and Indian house crows *Corvus splendens*.

Among Australia's most notorious exotic vertebrate species are the common starling *Sturnus vulgaris*, feral cat *Felis catus*, European rabbit, red fox, feral pig *Sus scrofa*, feral goat, house mouse *Mus musculus*, and black rat. All of these species are not only pests in Australia but around the world (listed among the 100 world's worst invasive alien species: www.issg.org/worst100_species.html). In all of these cases, the reason these pests have become well-known is likely due to their longstanding cultural associations with humans and their large, easily quantifiable, negative impacts on the recipient environments in the wild.

Native species can also become pests, in exactly the same way as exotic species (Richardson and Ricciardi 2013); when a perturbation to the host system causes a change in its interaction with its environment. Although perturbations can be natural in origin, such as an extreme weather event or earthquake, they are more often a direct or indirect result of human activity. One global example is habitat destruction and fragmentation (Brooks et al. 2002; Fahrig 2003; Hanski 2011). In Australia, large tracts of forest and woodland have been converted to pasture, agriculture, and silviculture. Land clearance opens the habitat for grasses that are exploited by large-bodied herbivores whose populations thereby increase and can negatively affect pastoral yields. In Australia, pastoralism has also been associated with a large increase in artificial point water sources (i.e., dams). This increase in grasslands and water availability is linked to significant increases in macropod (kangaroo and wallaby) populations and densities, and many native species are regarded as pests, and have been since European arrival (Boom et al. 2012).

Sometimes, a native species is introduced outside its current range as part of a conservation or 'translocation' program designed to safeguard it from a perceived threat, and ultimately from extinction. For example, the koala *Phascolarctos cinereus* was introduced to Kangaroo Island, South Australia, in the 1920's, and subsequently caused widespread habitat degradation and threatened regionally important populations of the manna gum *Eucalyptus viminalis ssp. Cygnetensis* (Masters et al. 2004). In Tasmania, 22 superb lyrebirds *Menura novaehollandiae* were introduced to two locations, between 1934 and 1949, in an effort to save the species from the perceived threat of foxes and habitat loss on the rest of the Australian mainland (Sharland 1952). Recorded sightings indicate lyrebirds are now established throughout much of southwest Tasmania, and the species could be having a significant impact on forest ecosystems through habitat modification (Mallick and Driessen 2009). In other cases, a species has been transported for people's pleasure. The rainbow lorikeet *Trichoglossus haematodus*, native to Eastern Australia, was introduced into the wild in Perth, Western Australia, during the 1960s. From fewer than 10 escaped (or released) birds, the population is now distributed over a large part of the metropolitan area and may number over 20,000 birds. The Western grey kangaroo *Macropus fuliginosus* was favourably introduced to Granite Island, South Australia but was later eradicated due to its negative impacts on the island's population of little penguins *Eudyptula minor*(Robinson et al. 1996).

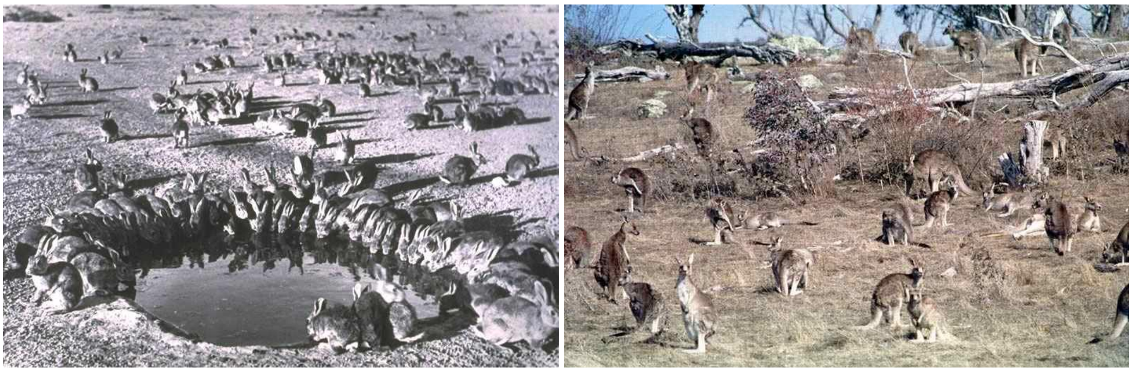

Figure 1: Examples of vertebrate herbivore 'plagues' in Australia: European rabbit, an exotic pest (left panel), and the Western grey kangaroo (right panel), which can be a native pest; particularly in regions of intensive land clearance and pastoralism. Copyrights: CSIRO and North Canberra Community Council.

1.3 Impacts of vertebrate pest species in Australia

1.3.1 Environmental impacts

Australia has one of the world's worst records for recent species extinctions, particularly exemplified by mammal extinctions. The latter account for almost one-third of all mammal extinctions in the last 500 years, and more species than anywhere else in the world (Burbidge et al. 2008; Figure 2). This infamous extinction record is due, at least in part, to Australia's high diversity (past and present) of small-bodied, long-lived, and relatively slow-reproducing endemic species that are particularly susceptible to extinction (Cardillo 2003). For the most part, however, it can be attributed to a series of enduring perturbations that began with human overexploitation of megafauna (Johnson 2006) and worsened with the arrival of the Europeans.

Following European settlement, acclimatisation societies zealously introduced exotic species to shape the environment into something more similar to their homelands. At the same time, the new settlers were clearing huge areas of native habitat (up to 40%) to make way for agriculture and pastoral practices (Bradshaw 2012). These large-scale perturbations had profound consequences on native biodiversity, causing a dramatic increase in extinction threat.

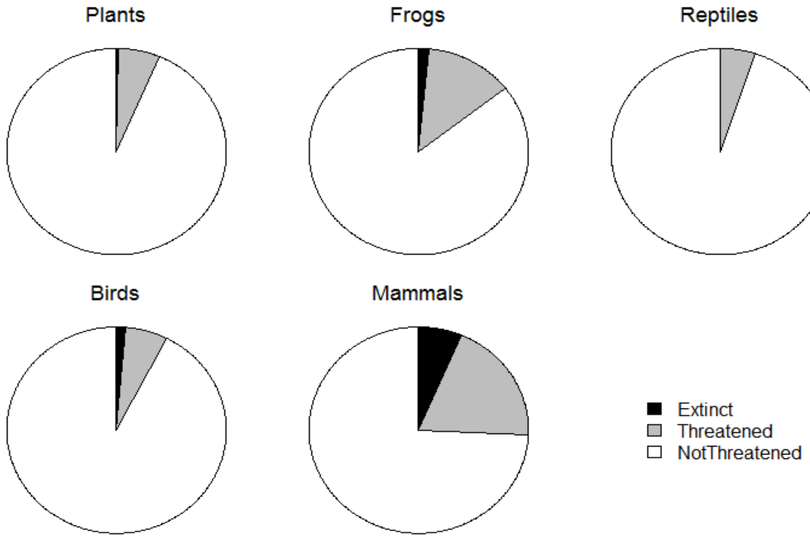

Figure 2: Australian extinctions by taxa. Proportionally more mammal species are threatened, and have already been lost through extinction, relative to any other taxa. Furthermore, the majority of mammal extinctions have been medium-sized species (the so-called 'critical weight range' between 35 and 5,500g (Johnson and Isaac 2009), including bandicoots, possums, quolls, small macropods (wallabies, potoroos, bettongs and rat-kangaroos) and large native rodents.

The extent to which these individual perturbations (habitat destruction and introduction of exotic species) are responsible for Australian biodiversity loss is in debate (Berglund et al. 2013; Clavero and García-Berthou 2005; Gurevitch and Padilla 2004), and is complicated by the inescapable fact that these events have already passed and often went unrecorded. There have been some attempts to reconstruct extinction events to quantify the likely cause of extinction for individual species, (e.g., the Thylacine *Thylacinus cynocephalus*; Prowse et al, 2013), but these efforts are labour-intensive and subsequently rare. Nevertheless, there are observations for which the temporal congruence of an introduction of an exotic species, and subsequent extirpation of a native species, suggests that the two are linked. Black rats arrived on Lord Howe Island in 1918, and are strongly implicated in the extinction of five species of endemic birds, two species of plants, and at least 13 species of invertebrates (Wilkinson and Priddel 2011). The arrival of the cane toad and domestic cats to Vanderlin Island in the Sir Edward Pellew island group in the Gulf of Carpentaria likely caused the local extirpation of a native quoll *Dasyurus sp.* population and brush-tailed phascogales *Phascogale tapoatafa*, brush-tailed rabbit-rats *Conilurus penicillatus* and canefield rats *Rattus sordidus* (Taylor et al. 2004).

Anecdotal records of local extirpations due to exotic vertebrate pest introductions are compelling but other factors are difficult to rule out entirely. In contrast, exclusion experiments offer a measurable and replicable method to assess the impacts of vertebrate pests on native flora and fauna, particularly non-volant vertebrate predators (Salo et al. 2010; Salo et al. 2007). Coupled with species reintroductions, some long-term exclusion experiments have shown that self-sustaining populations can persist in the absence of exotic vertebrate predators and herbivores, particularly feral cats and red foxes (Somers and Hayward 2012), and European rabbits.

However, the removal of any one pest species is not straightforward (Rayner et al. 2007). For example, Risbey et al. (2000) showed that removing both cats and foxes had a positive impact on small mammal populations around Shark Bay in Western Australia, but that removing only foxes resulted in increased small mammal declines. This was presumably due to heightened cat predation; an interaction through their release from interference with foxes. In fact, the removal of a single predator species can drive an increase in its prey species, providing the opportunity for competition with species that were previously unaffected by the original predator. This is an example of a so-called “secondary effect” (Section 5.1.4), and was claimed to be the case during the successful eradication of cats from the sub-Antarctic Macquarie Island in 2001 (Bergstrom et al. 2009); but also see Dowding et al. (2009). It was estimated that at least 4,000 rabbits per year re-entered the island ecosystem after the last cat was killed, although it is unclear whether this had more to do with the irregular supply of *Myxoma* virus than simple release from cat predation (Dowding et al. 2009). Regardless, increased rabbit herbivory, following the cat eradication, caused substantial damage at both local and landscape scales, including changes from complex vegetation communities to short-grazed lawns or bare ground. Increased erosion and rabbit tunnelling was also observed to destroy petrel burrows and further increase predation on petrels by skuas *Catharacta longbergii*. It is worth noting that eradication techniques for rabbits (and rodents) on an island the size of Macquarie were untested when cat eradication was deemed necessary, and that the successful cat eradication had substantial and rapid benefits for a range of seabird species (Dowding et al. 2009). Subsequently, aerial baiting of Macquarie Island for rabbits and rodents was completed in July 2011. At the time of writing this report no further rabbits had been killed (or detected) since November 2011. Rabbit hunting has continued and rodent detection dogs are deployed to confirm the presence (or absence) of rats and mice.

1.3.2 Economic impacts

Vertebrate pest species cause economic damage in a variety of ways, and some of the major threatening processes are listed in Table 1. Generalist pests, like feral pigs, cause damage to agricultural crops, depredate native plant and animal populations, and are thought to spread diseases, including tuberculosis, brucellosis, rabies, and foot-and-mouth disease to livestock and humans (Nentwig 2007). Exotic rodent populations reach extraordinary densities on both the mainland and offshore islands, where they are a social and economic pest, causing economic damage to standing and stored crops and impacting native species through predation and competition (McLeod and Norris 2004). A nationwide sample of horticultural growers suggested that birds are perhaps the single greatest burden to horticultural productivity, mainly by predating the fruit before it is harvested (Tracey et al. 2007). Apart from their cost due to direct damage, there is also a non-negligible cost incurred for researching optimum management techniques for all these species, and other vertebrate pests.

In Australia, McLeod and Norris (2004) estimated the cost of economic damages to agricultural production, accounting for the costs of mitigation, for 12 vertebrate pest species including major native (and naturalised) vertebrate pest species, kangaroos and dingoes. Their final estimate was $373.3 million per year. McLeod and Norris (2004) also estimated the environmental impact of vertebrate pest species at $345.8 million per year, and concluded that the house mouse had the greatest social impact within Australia. A similar exercise was attempted more recently by Gong et al. (2009). The results of the two studies are not directly comparable because the methods used to quantify economic damages were fundamentally different, as were the species considered. Nevertheless, Gong et al. (2009) estimated the cost of economic damages to agricultural production, again accounting for the costs of mitigation, for five exotic vertebrate pest species (foxes, rabbits, wild dogs, pigs and mice) and pest bird species (exotic and native) at $743.5 million per year. Gong et al. (2009) did not attempt to estimate the environmental impact of these species, citing insufficient data. One notable difference between these reports is that Gong et al. (2009), but not McLeod and Norris (2004), quantified the cost of economic damages by bird species, which accounted for over half of the total cost in agricultural (horticultural) losses.

The economic gains can occasionally outweigh the cost of economic damages for some vertebrate pest species. For example, recreational angling of exotic and native fishes in some Australian States, notably Tasmania, is widely promoted both nationally and internationally because of the revenue it generates (Ayres and Clunie 2010). This might, however, be an artefact of the difficulties in measuring the cost of economic damage caused by fish, not to mention their considerable environmental damage and loss of native small-vertebrate and macro-invertebrate biodiversity through predation (Gozlan et al. 2010).

Table 1. Some of Australia's most prolific vertebrate pests, with an example of their major damage or Listed Key Threatening Processes and the estimated annual cost of that damage on agricultural production in millions of Australian dollars (AUD).

Species		Damage or Key Threatening Process	AUD Cost[b]
Herbivore			
goats	*Capra hircus*	Habitat destruction & competition with livestock	4.23
rabbits	*Oryctolagus cuniculus*	Soil erosion & competition with livestock	88.11[d]
kangaroos	*Macropus sp.*	Competition with livestock	27.48
Omnivore			
mice	*Mus domesticus*	Spoiling & removal of grain	23.11
rats	*Rattus rattus*	Depredation of native animals	NA
pigs	*Sus scrofus*	Soil erosion and crop damage	100.00
Carnivore			
foxes	*Vulpes vulpes*	Depredation of livestock	17.50
cats	*Felis catus*	Depredation of native animals	NA
dogs[c]	*Canis lupus sp.*	Depredation of livestock	48.30

[a] as identified and listed in Australia's Environment Protection and Biodiversity Conservation Act 1999; see www.environment.gov.au/cgi-bin/sprat/public/publicgetkeythreats.pl

[b] direct cost due to agricultural losses taken from McLeod and Norris (2004)

[c] includes feral wild dogs and dingoes

[d] Gong et al. (2009) concluded that the disaggregated impacts by industry and State were both highest for rabbits, in the beef industry and State of Queensland respectively. Their total estimated annual cost for the overall loss in agriculture, including horticulture, to rabbits was $206 million, 2.5 times more than McLeod and Norris (2004).

For some vertebrate pest species, we have very little information on their estimated damage and cost. This is principally because their impact is largely environmental rather than directly affecting economic activities, which is considerably more complicated to calculate. Pimentel et al. (2007) estimated the economic impact of Australia's 18 million feral cats at over USD$540 million per year, assuming that the value of each bird depredated was USD$30. The dollar value attributed to an individual bird is, however, quite subjective and McLeod and Norris (2004) alternately estimated that 18 million cats, kill eight birds per year valued at AUD$1 per bird, giving a total economic impact of feral cats in Australia of AUD$144 million.

Given that exotic pests impact the Australian economy at all levels, from individuals to the country's national and international performance, the capacity to measure the cost of economic damage should be better developed (although see Gong et al. (2009) for a detailed discussion). Perhaps it is for this reason that those species, which cause the largest perceived direct economic damage, e.g., feral rabbits, dogs and pigs, have more often been the focus of greater management efforts.

1.3.3 Social impacts

Vertebrate pest species also have significant social impacts, although these are considerably more difficult to quantify. At a national level, the main social impacts of pest species are likely to be indirect, and flow from economic impacts and, to a lesser extent, environmental impacts. A study of the social cost of invasive animals to Australia (Fitzgerald 2009), demonstrated that impacts include psychological stress (e.g., anxiety, frustration and depression), due to the loss or damage of assets (e.g., livestock or infrastructure); fear of physical attacks on livestock, family members and pets; and trauma associated with vehicle accidents involving pest animals. Alternately, some exotic pest species were also cited for bringing positive social impacts to people (e.g., some Australians enjoy having cane toads as pets, or seeing rabbits in the landscape, or hunting and fishing exotic species for recreation). Assessing the social impact of vertebrate pests is complicated and requires the allocation of funds to scientific research, which should be replicated, long–term, and conducted across relevant scales (Braysher 1993).

1.4 Mitigating pest impacts: control or eradication

Ultimately, the goal of any vertebrate pest management is to reduce their environmental, economic and social impacts. Subsequently, most management programs attempt to minimise the number (or density) of the vertebrate pest, assuming that this will result in a reduction in their impact. However, it should be noted that a reduction in numbers does not always result in a reduction in impact, and may even result in a new (previously unforseen), or greater, impact. There are two options to mitigate the impacts of vertebrate pests: *control* or *eradication*.

1.4.1 Control of vertebrate pests

Control (including containment) is generally a long-term strategy to reduce (rather than remove) the impact of a pest species, usually by lowering their numbers, or restricting their geographical range or reproductive output. Because it requires a long-term effort, control is generally expensive and, over time, prone to chance events or inconsistency in funding.

A wide range of control methods are, or have been, used in Australia, including exclusion fences, poisoning, hunting or trapping, vegetation burning, release of infectious diseases, immunocontraception, and even biocontrol (through the release of yet another exotic species). Many of these control attempts are vast in scale and cost and some are globally well-known.

Control is the only option if it is not feasible or desirable to eradicate the target species (either because of the cost or social/environmental impact). While control can be effective it comes at a long-term cost and suffers from varied levels of success. Exclusion fences have been widely used to control pest species in Australia and include the century-old “dingo fence” and “rabbit-proof fence” (Breckwoldt 1988; Coman 2010). The dingo fence is the world’s longest fence at 5,614 kilometres (km) and was designed to exclude dingoes from south-eastern Australian States from which they had been largely exterminated to protect sheep grazing. The rabbit-proof fence consists of multiple structures stretching over 3,253 km and separates pastoral lands of Western Australia from rabbits (and other herbivorous pest vertebrates) in the east. The first fence was built in 1885, and between 1885 and the outbreak of World War 1 enough netting fence was erected in Australia to encircle the earth eight times (Coman 2010). The dingo fence is still maintained today and new fences are being constructed, including a 490 km proposed extension around Esperance, Western Australia. Maintenance of the rabbit-proof fence stopped following the introduction of *Myxomatosis* virus into the Australian rabbit population in the 1950s. Although neither fence has been 100%

effective at excluding the target species, they have reduced pest species densities in the protected areas; even if the pest species were not those targeted. The rabbit-proof fence was better at excluding pigs, goats and emus than rabbits. This reduction in pest species densities, however, came at a significant financial cost: at one time the rabbit-proof fence was patrolled by 120 men and 600 camels, horses, and donkeys. The dingo fence is still patrolled today and there are at least 23 fulltime staff dedicated to maintaining it.

Cane toads are a prolific vertebrate pest species in Australia that were introduced to northern Queensland in 1935 to control the sugar cane canegrubs (larvae of melolonthine scarabs); particularly *Dermolepida albohirtum*, for which other control methods had been deemed ineffective (Robertson et al. 1995). From 102 toads, released around Cairns and Innisfail, there are now estimated to be over 2 million spread from the border with New South Wales, through the Northern Territory and recently into Western Australia (Phillips et al. 2006). There is no doubt that the environmental impacts of the cane toad are far greater than those of the canegrubs (which are still a pest). The economic damage caused by the toad is likely to be considerable, particularly the significant funds invested for research into its effective management. The cane toad continues to be the subject of long-term control efforts.

1.4.2 Eradication of vertebrate pests

Eradication is the complete and permanent removal of the target pest species from a defined area, and within a defined time period (Bomford and O'Brien 1995). This definition is important because without constraining management actions to a specific area and time the campaign reverts to a control strategy. If effective eradication resources are not available eradication should not be attempted. To knowingly implement an ineffective eradication attempt is detrimental to the successful funding of other eradication programs.

In their landmark review, Bomford and O'Brien (1995) highlighted several factors that were critical for eradication to be successful (see also Parkes and Panetta, 2009). Rather than focusing on the physical attributes of locations where eradications are conducted, they focused more on the eradication procedure, as it can apply to any species, at any location, and across multiple methods. They listed three essential and three highly desirable criteria that if met would maximise the chance of eradication success. Their work has become a cornerstone for eradication programmes and literature.

Bomford and O'Brien (1995) outlined three essential criteria that *must be met* if a planned eradication is to be successful:

(**Criterion 1**) *the rate of removal must exceed the rate of increase at all population densities*;

(**Criterion 2**) *immigration must be zero* (i.e., no chance of re-invasion); and

(**Criterion 3**) *all reproductive animals must be at risk to the eradication method*.

If any of these three criteria cannot be met, then the eradication will not succeed and long-term control is the only management option. Parkes and Panetta (2009) added an additional essential criterion. Before proceeding with an eradication *there must be no net adverse effects*. This is particularly pertinent if the adverse effects on non-target species of the eradication are unacceptable, or if the (economic, social, or environmental) consequences of eradication of the pest outweigh the benefits.

Further to the above essential criteria, Bomford and O'Brien (1995) outlined three highly desirable criteria in order for a planned eradication attempt to be successful:

(**Criterion 4**) *Individuals must be able to be detected at low densities*.

If animals cannot be detected at low densities there is no way to measure whether ongoing eradication efforts are effective, and no way to determine if (or when) the eradication has been successful.

(**Criterion 5**) *Discounted benefit-cost analysis needs to favour eradication over long-term control.*

It is possible that the immediate cost of an eradication may exceed the long-term costs of reducing a population to low-numbers, and this must be formally considered.

(**Criterion 6**) *A suitable socio-political environment is present.*

Conflicting community or administrative goals (or legal barriers) can hinder eradication attempts. Reliable information about the effects of target species on production systems or environmental resources is necessary to build community support and create the political will required to achieve eradication success.

Where eradication of an exotic pest is feasible, then it is always preferable over control because: (1) it reduces the pest species impact to zero rather than minimising it; and (2) it is a shorter-term strategy with a definite end-point, and therefore, compared to control and containment, it is usually less expensive, although the immediate cost can still be considerable. The preference for eradication over control has been formally acknowledged for invasive species by the Convention on Biological Diversity, which states that after "*failing to prevent the arrival of a species, eradication should be considered the next management action, which if infeasible or unsuccessful, should be followed by long-term control*" (Genovesi 2007).

The majority of past eradication attempts, around the globe, have been for vertebrate pest species and have occurred since the 1930's when eradication to protect biodiversity became popular. Prior to this time, eradication was largely to protect crops or for sanitation (Genovesi 2007). Since then, there have been over 630 successful eradications of established invasive species recorded worldwide. These eradications have been achieved using a subset of the methods used for long-term control, the most common of which include poisoning, trapping and hunting. As with long-term control, a combination of methods can be more effective than using any single method on its own (Genovesi 2007). This could be because the population becomes accustomed to a particular method and learns to avoid it (Bischof and Zedrosser 2009), such as individuals that become "trap shy". In other cases, a variety of techniques and applications is necessary to put all individuals in the population at risk; Criterion 3 (Bomford and O'Brien 1995).

To date, most successful eradications of (established) vertebrate pest species have been on islands; there is no record (yet) of the successful eradication of a widespread vertebrate pest species from any continent on Earth (Bomford and O'Brien 1995). Although many exotic vertebrate invasions have been detected and extirpated before they could increase and become widespread. This observation raises a number of interesting questions. Firstly, have there been an equal number of eradication attempts on continents, or have most been on islands? Islands represent a known area of isolated habitat/environment and are therefore considered more feasible for eradication, i.e., high probability and low risk. Islands are also hotspots of endemism and often refugia for threatened species so may be valued more for immediate management and eradication. Secondly, does the probability of eradication success decline with an increase in the area or the topographic complexity of the region to be eradicated?

1.5 Studies of eradication success

There have been several studies of global eradication success, although most have traditionally been qualitative, as opposed to quantitative. Myers et al. (2000), Clout and Veitch (2002), Simberloff (2003), Nentwig (2007), and Parkes and Panetta (2009) all review factors that may determine the success or failure of eradication attempts for any species (see also Parkes 2006). Although they all draw similar conclusions, some emphasise one factor over another. For example, Simberloff (2003) highlighted the need for early detection and action, which if delayed is thought to render eradication less likely to be successful, particularly due to increasing cost, among other factors.

A large collection of studies has focused on eradications of particular taxa or particular locations, e.g., exotic mammals on islands. Some of these studies contained quantitative summaries but none have included formal statistical analyses. Genovesi (2005) reviewed eradication attempts in Europe and found that the large majority were on islands. Courchamp et al. (2003) comprehensively reviewed the literature on mammal eradications on islands. In separate studies, Nogales et al. (2004), Campbell and Donlan (2005) and Towns et al. (2011) reviewed eradication attempts on islands for single species; feral cats, feral goats, and black rats respectively.

Clout and Russell (2006) examined exotic mammal eradications on New Zealand islands and found that there was a substantial increase in the number of successful eradications during the 1980s and 1990s. They concluded that the most significant change was the ability to eradicate rodents from increasingly large islands (to over 11,000ha), using aerial poisoning techniques. Hess and Jacobi (2011) described the history of mammal eradications in Hawai`i and the United States associated islands of the Central Pacific. While whole-island eradications have been achieved in a number of cases, in other cases (on the largest islands), pest-free areas were only achieved through the maintenance of predator exclusion regions. Aguirre-Muñoz et al. (2008) and Aguirre-Muñoz et al. (2011) examined the ecological outcomes attained after 49 populations of 12 invasive mammals were eradicated from 30 Mexican islands. They concluded that the eradication of all exotic vertebrates from Mexican islands was a viable and achievable strategic goal. Oppel et al. (2011) reviewed the operational challenges associated with eradications of exotic mammals on islands with permanent human populations. Because of the additional social complications when performing eradications on inhabited islands (e.g., community opposition on ethical or cultural grounds, habitat refuges on private property, increased opportunity for re-invasion with private transport), the authors recommended a close collaboration between island communities, managers, and social scientists from the inception of an eradication campaign.

Other non-vertebrate eradications have also been studied in this manner, and their approaches and conclusions are of interest regardless of the taxa. Mack and Lonsdale (2002) and Rejmánek and Pitcairn (2002) reviewed the factors thought to be important for the eradication of plants. Brockerhoff et al. (2010) studied the costs and benefits of invasive forest insect eradications, and correlates of their success. A more formal statistical analysis of determinants of non-vertebrate eradication success was recently undertaken by (Pluess et al. 2012a; 2012b). The analysis in the latter study used a larger dataset because it allowed for missing values, and therefore included a wider range of explanatory variables. The former study found that area infested was an important factor determining eradication success: smaller areas were more likely to be successfully eradicated. They also found that preparedness was secondarily important, which they defined on a categorical scale, whereby being armed with the adequate knowledge and funding to tackle an invasion was associated with higher probability of eradication success (Pluess et al. 2012a). Among the more important factors in the latter study was whether the habitat was man-made (including greenhouses; an extreme man-made habitat in which eradication was most often achieved) and, again, the spatial extent of the infested area to be eradicated (Pluess et al. 2012b). It is interesting to note that failed eradication attempts made up approximately 50% of their dataset (Pluess et al. 2012a; Pluess et al. 2012b).

To date, we do not know of any study that has conducted a formal statistical analysis to determine the factors affecting eradication success for vertebrate pests on islands (or mainland).

2 AusErad: the Australian vertebrate pest Eradication database

2.1 Compiling the database

With support from federal and State government agencies, we compiled a database of all known vertebrate pest eradication attempts from Australian offshore islands and mainland States and Territories (including Tasmania). Known simply as AusErad, the database is a compilation of the Australian entries in the Database of Island Invasive Species Eradications (DIISE; www.islandconservation.org/tools/?id=67) and the Department of Sustainability, Environment, Water, Population and Communities (SEWPaC; www.environment.gov.au) pest animals on offshore islands database, supplemented with accounts from other databases (e.g., ISSG Global Invasive Species Database; www.issg.org), primary literature, e.g., (Veitch and Clout 2002; Veitch et al. 2011), and the expert opinion of Commonwealth and State national parks rangers and wardens (see Ackowledgements).

AusErad includes information on three main aspects of vertebrate pest eradication attempts: (1) physical and biological factors at the location of the eradication attempt; (2) details of the vertebrate pest population eradicated; and (3) details of the eradication itself, i.e., methods and cost. A list of the current information fields in AusErad are presented in Appendix A. This list is provided as a standard for the future recording of information on vertebrate pest eradication outcomes in Australia.

2.2 Reporting bias

It is well acknowledged, that there can be a bias in the reporting of eradication attempt outcomes, although there is some dispute as to the nature of the bias. Simberloff (2009) argued that successful eradication attempts are under-reported whereas unsuccessful eradication attempts are over-reported, reflecting the tendency of the media to focus on negative events. Alternately, it has been argued that unsuccessful eradication attempts go largely unreported because the agencies responsible for undertaking the programme do not want the lack of success to be publicised, potentially jeopardising their reputation and ability to attract already limited funding (Csada et al. 1996).

AusErad records a very low proportion of unsuccessful eradication attempts: only 11.7% of all reported vertebrate pest eradication attempts (recording either a known success or failure) were determined to be unsuccessful. It is not certain, however, that this low proportion of eradication failures is due to reporting bias. Howald et al. (2007) reported 81% and 95% of eradication attempts of house mouse and Norway rat *Rattus norvegicus* on islands were successful. Aguirre-Muñoz et al. (2008) and Lorvelec and Pascal (2005) found that the outcomes, for Mexican islands (42 out of 44; 95%) and French Overseas Territories (28 out of 34; 82%), of island eradications of exotic mammals had also been impressively rewarding.

Similarly, the DIISE reports that approximately 90% of recorded vertebrate eradication attempts have been successful.

Given that many of the eradication attempts reported in Howald et al. (2007) and others (including Nogales et al. (2004); Campbell and Donlan (2005) and Oppel et al. (2011)), are also recorded in DIISE and that the DIISE was one of the main sources of data for AusErad, we acknowledge that there is a degree of circularity here. In an attempt to minimise potential under-reporting of unsuccessful eradication attempts, we established an “Australian Eradication Attempts” Google Group and invited 38 experts of Australian eradication attempts to comment, question and discuss topics from biases in AusErad to individual eradication attempts in an open and accessible forum. We concluded that the AusErad summary plots (and associated information) accurately reflected prevailing expert opinion on Australian eradication attempts. More details about the Google Group are provided in Appendix B.

2.3 AusErad: a brief overview

AusErad currently contains information on 650 vertebrate pest population eradication attempts; 354 from 188 offshore islands and 296 from various locations on the Australian mainland (including Tasmania). As expected, the number of reported eradication attempts has increased over time, on both islands and mainland, and this appears to be the case for both successful and failed eradication attempts (Figure 3).

Eradication attempts cover a wide range of exotic vertebrate species, from guppies *Poecilia reticulata* to Indian peafowl *Pavo cristatus* and to feral horses. In total, there were records of an eradication attempt for 68 different species. The majority of these were mammals, particularly on islands, followed by fish on the mainland (Figure 4).

Finally, it seems that Western Australia has undertaken the highest number of island eradication attempts, although many of these are in the same island group from a single campaign. On the mainland, the number of eradication attempts was similar across all States (Figure 5).

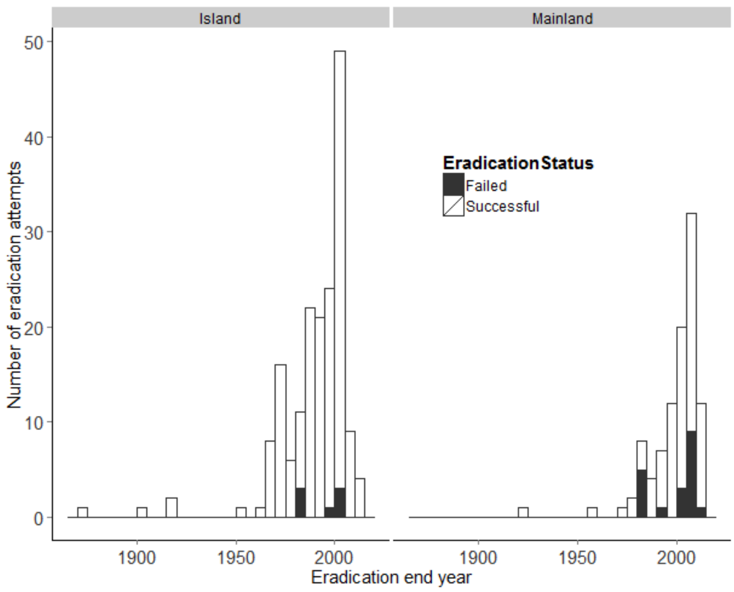

Figure 3: Histograms of the numbers of successful and failed eradication attempts (species by location) over time, separately for islands and mainland.

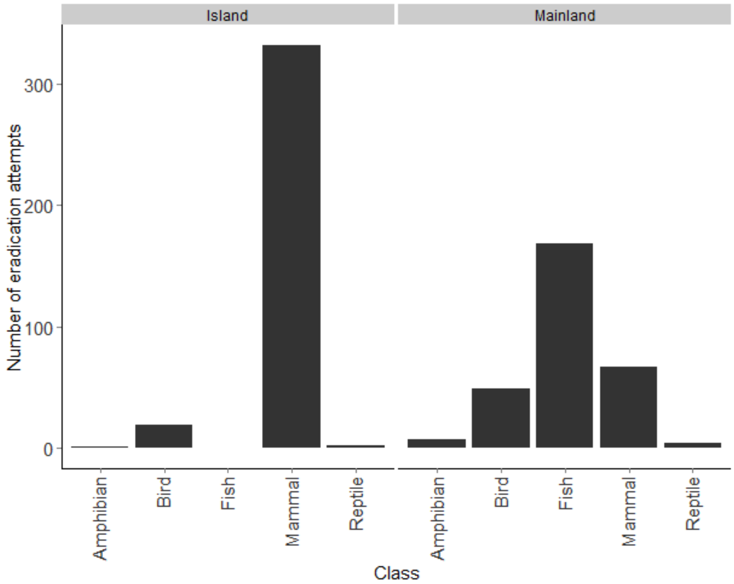

Figure 4: Histograms of the numbers of eradication attempts of each class of vertebrate pest, separately for islands and mainland.

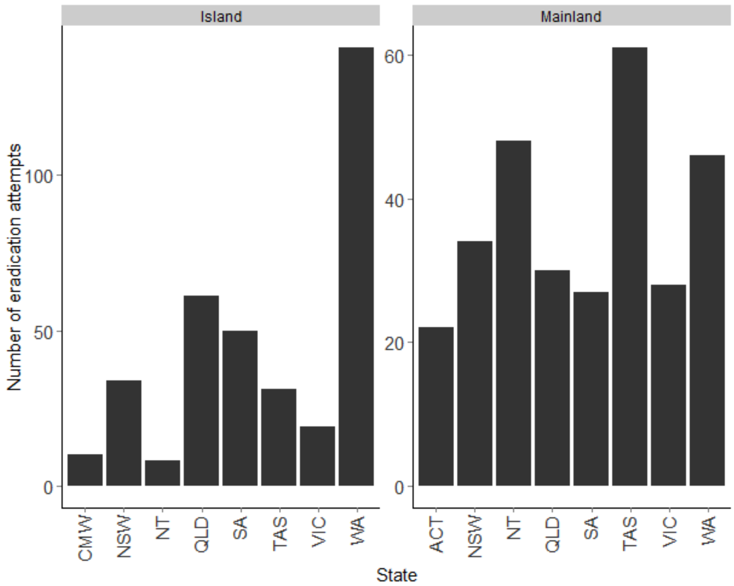

Figure 5: Histograms of the numbers of eradication attempts in each State, separately for islands and mainland. Note: all Australian Commonwealth Overseas Territories (CMW) are islands.

3 Island eradication attempts

Due to their remoteness, and long periods of evolutionary isolation, islands are hotspots of endemicity, which means that although they constitute a small percentage of the earth's total area, they contribute disproportionately to global biodiversity (MacArthur and Wilson 1967; Myers et al. 2000). As such, islands are at considerable risk from the current global extinction crisis, including from the negative impacts of invasive vertebrates (e.g., Courchamp et al. (2003); Blackburn et al. (2004); Harris (2009); and Towns et al. (2011)). Removing invasive vertebrates from islands is therefore considered the most effective way to protect and restore island ecosystems and prevent extinctions (Donlan and Wilcox 2008). Accordingly, the majority of eradication attempts have been made on islands (with over 630 successful eradications recorded from offshore islands worldwide) and the majority of these were to eradicate vertebrate pest populations (Krajick 2005).

Australia has over 8,300 islands, the majority of which are defined as offshore, i.e., not in inland waters (www.ga.gov.au/). These offshore islands range in size from Melville Island (Northern Territory) at over 5,700 km^2 to Channel Rock (Western Australia) at less than 1 km^2. Some islands are connected to the mainland at low tide whilst others (Australian Overseas Territories) are >1,600 km away. In many instances, offshore islands exist as part of an archipelago; a group of geographically proximate islands of shared origin. In most cases, the islands were born of plate tectonic processes or erosion of mainland Australia. In a few cases, the islands have a more unusual origin, such as Fraser Island (Queensland), which is the world's largest sand island.

Burbidge (1999) highlighted the current and future importance of islands to biodiversity conservation, prompting national agencies to particularly focus on eradication of vertebrate pest species from offshore islands. Today, Australia (together with New Zealand) is a world leader in vertebrate pest eradication attempts. Of Australia's 8,300+ offshore islands, vertebrate pests have been recorded on some 518 islands (Department of the Environment Water Heritage and the Arts (DEWHA) 2008) and their eradication has been attempted on over one-third of those islands (Figure 6).

3.1 Studies of eradication success on islands

Howald et al. (2007) reviewed eradication attempts of invasive rodents from islands worldwide. However, they did not include subsequent eradication efforts of small rodent populations that reinvaded islands following a previous, successful eradication campaign, despite indicating that it is common on islands located close to a mainland source population. Howald et al. (2007) used four simple explanatory variables of success: island area, cost of eradication, method of baiting and the date (year) of the eradication attempt. They found that black rats had been eradicated from the most islands, followed by Norway rats, kiore *Rattus exulans* and mice, and that 78% of eradications were on islands <100 ha. Reported eradication failures were rare but highest for mice and lowest for Norway rats. In the past, bait stations was the most common method used, followed by hand delivery of bait and then aerial delivery. However, the latter method accounted for 76% of the total baited area over all programmes, and is now common for much larger island areas (Broome 2009). Brodifacoum was the most commonly used poison.

Campbell and Donlan (2005) reviewed eradication attempts of goats from islands worldwide. They found that goats had been eradicated from 120 islands. With the development of new techniques, including the use of Judas goats (tagged individuals that 'give away' the position of feral conspecifics) and Geographic Positioning Systems, they found that the success of eradication attempts has increased dramatically, and suggested that island size is no longer a barrier to goat eradication success. Nogales et al. (2004) recorded a total of 48 cat eradications from islands almost exclusively less than 290 km^2 in size. The most frequently used methods were trapping and hunting with dogs, although these were often combined with poisoning.

More generally, Oppel et al. (2011) reviewed attempts to eradicate exotic mammals from islands inhabited by humans and domesticated livestock. They warned that eradication attempts would be susceptible to failure unless the eradication team engaged closely with people living on the island. Glen et al. (2013) emphasised the same point with an analysis of a database of 1,224 successful eradications of invasive plants and animals on 808 islands. They also emphasised the importance of targeting multiple invasive species simultaneously, wherever feasible.

New Zealand has a long tradition of successfully eradicating invasive species from small offshore islands. Towns et al. (2011) reviewed eradications of pest mammals from at least 100 offshore islands around New Zealand, covering a total area of around 45,000 ha. They concluded that "*eradications of invasive mammals are aggressive conservation actions that can have wide benefits for biodiversity but can also be controversial, technically demanding and expensive*".

3.2 Explanatory variables of eradication success

We gathered ancillary data that was believed *a priori* to explain the probability of eradication success. The explanatory variables, their expected effect on eradication success, and the sources from which they were collected are listed in Table 2. We note that these variables do not constitute an exhaustive list of all possible traits influencing the success (or otherwise) of a vertebrate pest eradication but are instead the putative variables for which the majority of data were available. A more detailed justification for the inclusion of each explanatory variable and the details of how they were measured or calculated is presented below.

Table 2. Explanatory variables that were either collated or calculated for the analysis of eradication success on the islands in the AusErad database. (Tr = Transformation of the variable that was used in the subsequent analyses)

Variable	Units	Tr	Source
Island name & coordinates	NA	NA	Geonames database [1]
Region name	NA	NA	2012 Times World Atlas, primary literature, expert knowledge
Island size	km^2	log10	GEODATA TOPO 250K Series 3 [2]
Elevational range	5-100% range	sqrt	1s (30m) and 9s (250m) Digital Elevation Models [2]; 9s (250m) bathymetry [2]
Variation in ruggedness	coefficient of variation	sqrt	Calculated from Elevation
Island distance from mainland	Km	log10	Calculated from GEODATA TOPO 250K Series 3 [2]
Island group distance from mainland	Km	log10	Calculated from GEODATA TOPO 250K Series 3 [2]
Island distance from source island	Km	log10	Calculated from GEODATA TOPO 250K Series 3 [2] & SEWPaC pest animals on offshore islands database [3]
Time present on island	Years	NA	AusErad sources and primary literature

[1] www.geonames.org [2] www.ga.gov.au [3] www.environment.gov.au

3.2.1 Island name and region name

All islands were assigned a name consistent among databases and assigned to a region representing their common biogeography (biodiversity and human change) and management. Island names were cross-referenced with a number of sources because they were used to link AusErad to the Geosciences Austalia (GA) and SEWPaC databases.

An increasing awareness of the importance of re-invasion risk (the chance that an invasive species will reinvade an island from a nearby source population) has prompted managers to undertake eradication attempts on groups of islands simultaneously (Harris et al. 2012; Russell et al. 2009). In addition, it was expected that islands within the same group would be more likely to be managed by the same authority and more similar to one another (environmentally and ecologically) than to islands outside the group. For these reasons, we assigned each island to a region name, loosely defined as the collective name given to it and its surrounding islands.

In most cases, the region name was an archipelago name. For example, the Montebello archipelago in NW Western Australia encapsulates 73 islands on which eradication has been attempted. We cross-referenced to which archipelagos islands belonged using the 13th Edition of the Times World Atlas (2011). In some cases, however, an island could not be assigned an archipelago name and, in cases where these islands were close to other islands, we asked experts to confirm that the islands were a group and the name of that group. If the island wasn't considered part of a group, then its singular name was used as the region name.

Overall, the majority of records in AusErad were for islands that were either not part of an archipelago, or were one of only a few islands in an archipelago (i.e., three or less) that had been subject to an eradication attempt (Figure 7).

3.2.2 Island area

Several studies have suggested that the successful probability of eradication is a negative function of the area to be eradicated, whereby it decreases as the area increases (Brockerhoff et al. 2010; Courchamp et al. 2003; Liebhold and Bascompte 2003; Mack and Lonsdale 2002; Pluess et al. 2012a). We therefore calculated area in km^2 for all islands in AusErad. AusErad contained island areas for many islands inherited from the source databases. The GA database was spatial and so the islands were represented by polygons with an area attribute. We cross-checked the AusErad areas with those in the GA database and whilst there was a strong positive relationship between them (linear model: log(AusErad area) = -0.164 + (0.975 * log(GA area)), Adjusted R^2 = 0.944), there were some discrepancies and so we used the GA islands areas for consistency. We cross-checked a random selection of these areas against estimates published in the 2012 Times World Atlas, and were satisfied with the consistency.

3.2.3 Island elevational range and ruggedness

Island elevation (in metres) was used to calculate two variables: (1) elevational range, a proxy for habitat complexity, and (2) variation in ruggedness, a measure of topographic complexity (Table 2).

Elevational range was predicted to affect the probability of eradication success because islands with a large range of elevations were expected to have a greater habitat complexity (Lomolino 2001) and therefore be increasingly difficult to eradicate pests from. Because of the irregular shapes of many islands, in addition to data uncertainty, there were occasions where an island overlapped with an open water cell with elevation below sea-level (i.e., <0 metres). We estimated island elevation using the range in elevation after removing the lowest

5% of values, i.e., using only the 5-100% percentile range. This ensured that the elevation value of smaller islands, with a higher edge-to-area ratio (which constituted the majority of cases), were not biased low. For the majority of islands (*c*.90%), we were able to extract elevation data from the Geosciences 1 second Digital Elevation Model (*c*.30m DEM). Islands further offshore were outside the extent of the 1 second DEM. For those islands (mostly Australian overseas Territories but also the Houtman Abrolhos islands), we calculated elevation using the Geosciences Australia 9 second (*c*.250m) DEM. Finally, in the case that these furthest offshore islands were very small so they were not present in the 250m DEM, we extracted data from the Geosciences Australia 250m bathymetry layer and calculated the absolute difference between the 5-100% percentile range.

Rugged terrain is predicted to complicate eradication whereby more rugged areas (including refugia) are more difficult to access and subsequently more difficult to bait or hunt (e.g., Parkes, et al., 2002; Veitch, et al., 2011). To account for this, we calculated the Terrain Ruggedness Index (TRI; Riley et al., 1999) for cell i as:

$$\mathrm{TRI}_i = \frac{\sum elev_i \cdot W}{8},$$

where

$$W = \begin{matrix} 1 & 1 & 1 \\ 1 & 0 & 1 \\ 1 & 1 & 1 \end{matrix}$$

is a matrix of all queen-case neighbours of cell i. In effect, the ruggedness measure for cell i is an average of all its immediate neighbours. We then calculated the coefficient of variation for island-wide mean ruggedness (TRI). Since TRI is an average measure, it was not expected to be biased by edge effects.

3.2.4 Island distance from the mainland and source island

Island distance from mainland, and island group distance from mainland, were calculated as explanatory variables of eradication success because it has been speculated that the further the island is from a location with an accessible port (and provision of supplies, transport and accommodation, etc.) the higher the probability that the eradication will not succeed. This is particularly relevant when the cost of unanticipated problems increases with distance (Donlan and Wilcox 2007).

The closer an island is to another population of the 'eradicated' pest species (e.g., a "source" island), the higher the reinvasion risk is expected to be. Eradication is defined as the *permanent* removal of a species from a specified location, i.e., if the island is reinvaded, then the eradication is said to have failed (or more specifically, because biosecurity measures have failed (Broome 2007), it is defined as a failed success). It is for this reason that reinvasion risk is a paramount consideration when planning island eradications (Harris et al. 2012; Russell et al. 2009). A reasonable proxy for reinvasion risk might be the distance from an area with a population of the same vertebrate pest species. For example, with knowledge that black rats can swim over 1.4km between islands, Burbidge (2004) highlighted the importance of eradicating black rats from all of the Montebello Islands in one large campaign.

For each island in AusErad, we calculated the distance (km) from any point on the island shore to the nearest point on the Australian mainland (including Tasmania) as the minimum Great Circle distance between the two localities. This was calculated using the `gDistance` function in the package `rgeos` for the R software environment for statistics and graphical computing (R Core Team 2013). In addition, we also measured the island group distance to the Australian mainland, taken to be the minimum island-to-mainland distance among all the islands in the group.

To locate the closest island with a source population of the same vertebrate pest species (a “source island”), we iterated through all islands with the same species in the DEWHA database, calculated the distance to the AusErad island and recorded the name and distance (km) of the closest island.

3.2.5 Period that a pest species is present on an island

A further variable with potential to influence eradication success is the length of time the population has been present on an island, i.e., the time between the date of introduction and the start of the eradication (Genovesi 2007; Myers et al. 2000; Pluess et al. 2012a; Simberloff 2009). Theory predicts, that the longer the population has been present, the more difficult it will be to eradicate because: (1) the population will have spread to all habitable areas of the island, including inaccessible areas; (2) individuals might have had time to adapt to the local conditions, thereby improving their resilience to eradication attempts; and (3) the cost of eradicating a population increases with population size, and can quickly become substantial (Myers et al. 2000).

We searched primary and grey literature and sought expert knowledge to obtain the species’ date of introduction and the eradication attempt start and end dates. From these data, we calculated the period a species was present on an island as the start year of the eradication minus the estimated year of the species’ introduction to the island. In many instances, neither our search nor expert knowledge yielded a date of introduction. Consequently, we substituted the date of introduction for the date that any one of the focal vertebrate pest species was introduced to the nearest island in the region (island group). Similarly, eradication start and end dates were rarely recorded, particularly start dates, which we could confidently assign for only 36% of the eradication attempts in AusErad. To overcome missing cases, we assumed that, when not recorded, the start date of the eradication attempt was 10 years after the island was denoted a protection status (D Priddel, *pers. comm.*).

3.3 Statistical analysis

All statistical analyses were conducted in the R software environment for statistics and graphical computing (R Core Team 2013). The ability of our chosen explanatory variables to estimate the probability of eradication success for islands in AusErad was tested using Generalised Linear Models (GLM). These models linked explanatory variables to the eradication outcome (a binary variable where 0 = a failed eradication attempt and 1 = a successful attempt), using the logit link function with binomially distributed errors. In GLM, the explanatory variables were fitted as fixed effects and species (see below) as separate intercept terms.

As a first step, we examined the ability of each of these explanatory variables (separately) to influence the probability of island eradication success. We generated confidence intervals around the univariate GLM model predictions by parametric bootstrap re-sampling (k = 200). In a second step, we built a global model-set that combined all of the putative explanatory variables. We tested the influence of these terms in multivariate additive models using the Akaike Information Criteria (Burnham and Anderson 2002) penalised for small sample size (AIC*c*).

Most of the eradication attempts recorded in AusErad were on isolated islands classified as the only member of their region (Figure 7). However, eradication attempts for the same species, on different islands within the same region, are likely to constitute non-independent events. For example, multiple populations could be eradicated as part of an archipelago-wide campaign undertaken by the same management authority using the same methods. To investigate the possible effect of non-independence, in the minority of within-region eradication attempts, we refitted the "top-ranking" five GLM models to 200 stratified bootstrap datasets to examine variation in their Relative Model Weights (from differences in their AICc values). For bootstrapping, we stratified within eradication outcome for each species in each region and resampled cases with replacement. This ensured that the GLM was refitted using the same number of eradication attempts for each species within each region.

We provide summary plots of all island eradication attempts collated in AusErad but we then restrict our formal statistical analysis to a subset of these species. Of the pest vertebrates established in Australia, the federal *Environmental Protection and Biodiversity Conservation Act 1999* (EPBC Act) identifies seven vertebrate pest species as threatening "*the survival, abundance or evolutionary development of a native species or ecological community*" by four main types of impact (www.environment.gov.au/): predation by feral cats, black rats and red foxes; competition and land degradation by European rabbits and feral goats; predation, habitat degradation, competition and disease transmission by feral pigs; and poisoning by cane toads. AusErad held sufficient records of eradications on islands to constitute a statistically viable sample size for six of these species; we excluded cane toads whose eradication attempts have been largely limited to mainland eradication attempts. Conversely, we included the house mouse in our analysis, for which we had sufficient records.

Our explanatory variables for the probability of eradication success were chosen because we had sufficient sample sizes to conduct a formal statistical analysis and they have been postulated in the literature as being important. Based on previous studies, we predicted the following relationships between explanatory variables and the probability of eradication success (p):

(i) Eradication success will decrease with an increase in island area (Pluess et al. 2012a), elevational range and variation in ruggedness (Parkes et al. 2002), distance of island and island group from the mainland (Donlan and Wilcox 2008), and time the species has been present on the island (Simberloff 2009); and

(ii) Eradication success will increase as the distance from a source population increases (Harris et al. 2012).

3.4 Results

3.4.1 Summary of island eradication attempts

At least 354 vertebrate pest population eradications have been attempted on 188 of Australia's 8,300+ islands (Figure 6). Most of these have occurred in Western Australia (WA), which had more than twice the number of recorded eradication attempts of the next highest States, Queensland (Qld) and South Australia (SA) (Figure 8). However, when grouped by Region, the number of island groups in which eradication has been attempted was similar for WA and Qld (Figure 8).

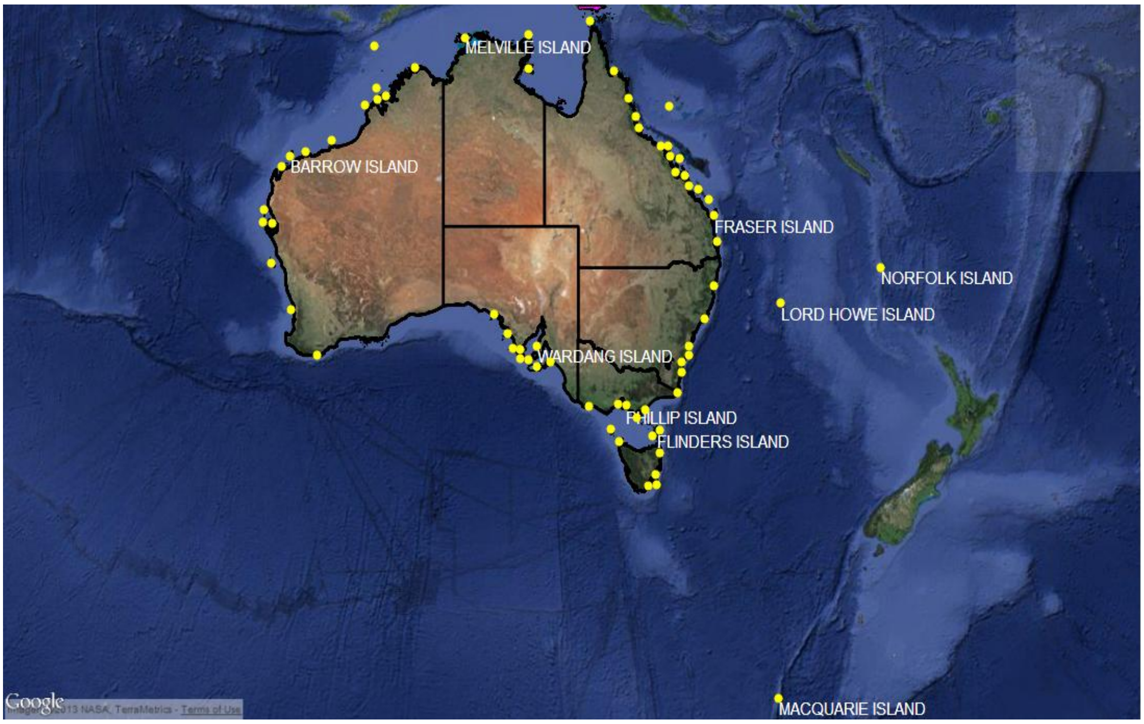

Figure 6: Map showing the locations of Australian offshore islands on which vertebrate pest population eradications have been attempted. Key islands discussed in the report are labelled.

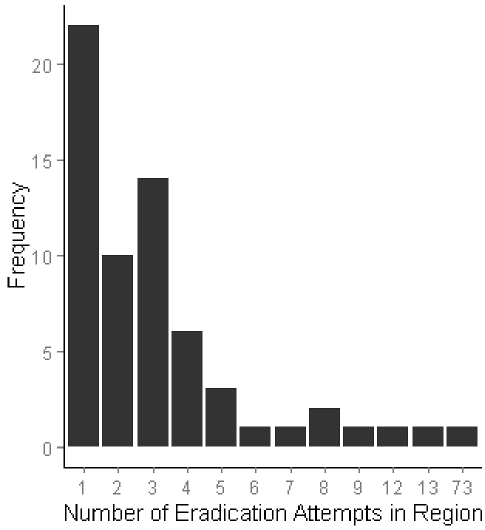

Figure 7: Histogram of the number of island eradication attempts for different 'Regions' (i.e., archipelagos or island groups).

Overall, reported eradication attempts on offshore islands have been successful (Figure 9): there have been just 25 of 335 (7.5%) unsuccessful attempts. Most eradication attempts were for birds and mammals (Figure 10), and it appears that all bird eradication attempts have been successful, although this might be affected by reporting bias and the level of post-eradication monitoring: the predicted risk of reinvasion for such volant species is high. The single reported amphibian and the two reptile eradication attempts were all successful.

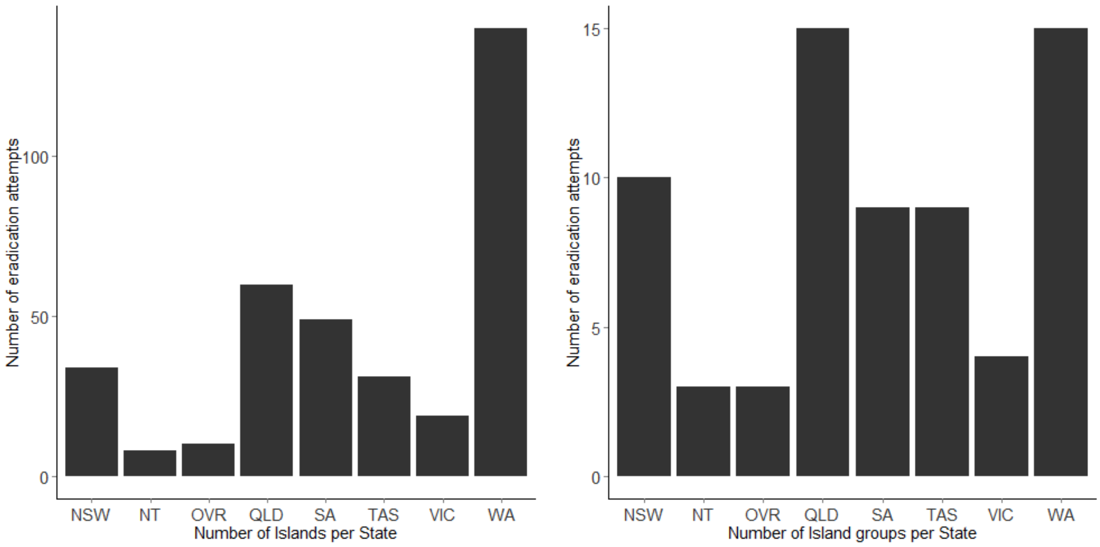

Figure 8: Frequency histograms showing: (i) the number of eradication attempts by Australian State; and (ii) the number of island group by State on which an exotic vertebrate pest population has been the subject of an eradication attempt.

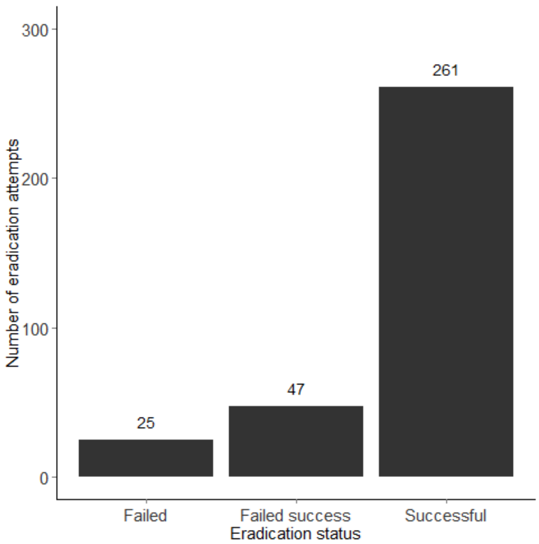

Figure 9: Frequency histogram summarising the success and failure of eradication attempts from Australia's offshore islands.

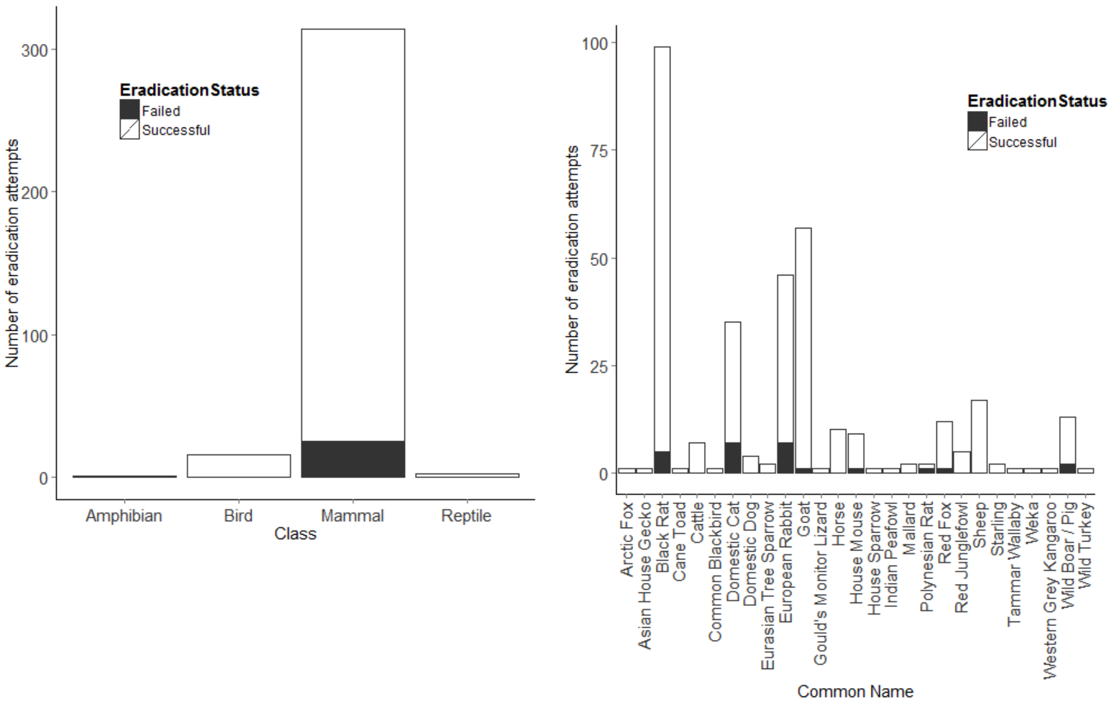

Figure 10: Histograms summarising the success and failure of eradication attempts from Australia's offshore islands by: (i) taxonomic Class; and (ii) individual species. Note that the number of amphibian eradication attempts is one (the Cane toad from Groote Eylandt), and it was successful.

Among mammal eradication attempts, the highest proportions of failures were for feral cat, European rabbit and black rats, followed by feral pigs (Figure 10). For some species, notably large ungulates, all eradications have been successful. Some of these species (e.g., cattle, horse, sheep) were present on the islands as domesticated stock and their successful eradication is likely, in part, because they were not true feral populations. Some of the other vertebrate eradication attempts were native species that had been introduced to offshore islands for biocontrol (Gould's monitor lizard *Varanus gouldii*) or pleasure (Western grey kangaroo).

3.4.2 Patterns in eradication attempts

For the remainder of the results, we focus on the following seven threatening vertebrate pest species, the number of eradication attempts for which are given in parentheses: black rat (n = 99), goat (n = 57), European rabbit (n = 46), feral cat (n = 35), feral pig (n = 13), red fox (n = 12), and the house mouse (n = 9). Total eradication attempts = 271.

The overall probability of successful eradication was highest for goats and black rats and lowest for feral cats, European rabbits and feral pigs (Figure 11). Pig eradication attempts have been conducted on the largest islands with high elevational ranges, while rat eradication attempts have been conducted on the smallest islands with low elevational ranges (Figure 12). Rat, mouse and pig eradication attempts were on more remote islands than for the other species, particularly compared to eradication attempts for red fox that were on islands nearest to the mainland. Rat eradication attempts tended to occur on islands closer to source islands than for other species, particularly pigs (Figure 13). This may be a consequence of rat eradication attempts occurring more often at the archipelago scale (island group) than for other species. Rats tended to be present on islands targeted for eradication attempts longer than other species, perhaps because they often arrived with very early explorers, before islands were invaded or stocked with other species (Figure 13). Mice, on the other hand, were present on islands for the shortest period before an eradication was attempted, perhaps because of widespread concern that if left unchecked, any invasion would spread and become prohibitively expensive to eradicate (Simberloff 2003).

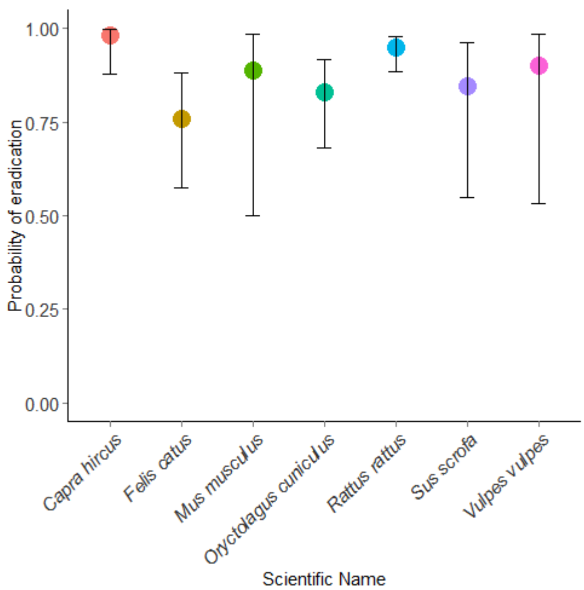

Figure 11: Differences in the probability of eradication success for the seven exotic vertebrate pest species that have experienced the greatest number of recorded island eradication attempts. Points are mean model estimates and error bars are 95% confidence intervals.

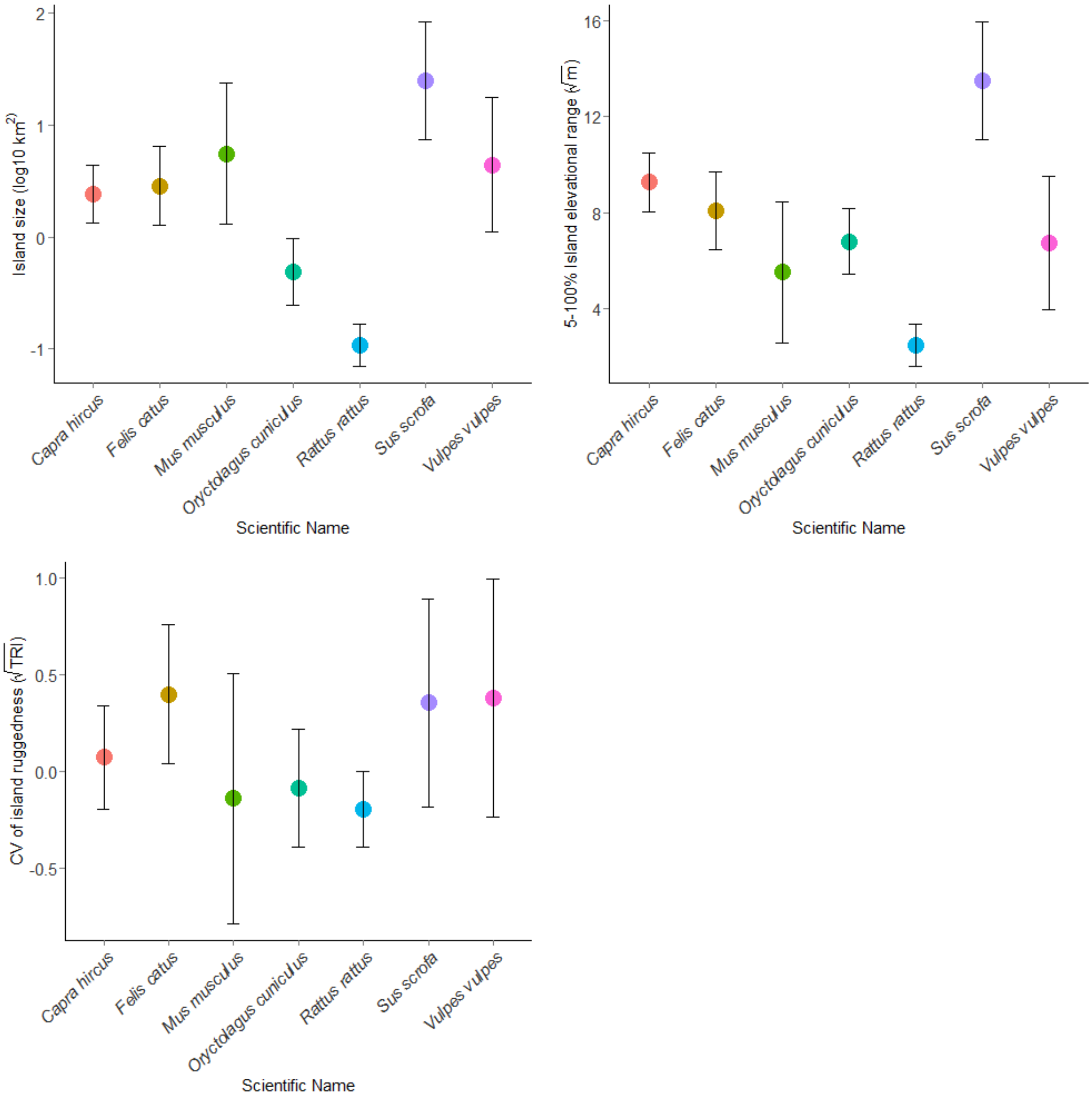

Figure 12: Differences in (i) the size, (ii) the elevational range, and (iii) the variation in ruggedness of islands on which eradications were attempted for each vertebrate pest species. Points are mean model estimates and error bars are 95% confidence intervals.

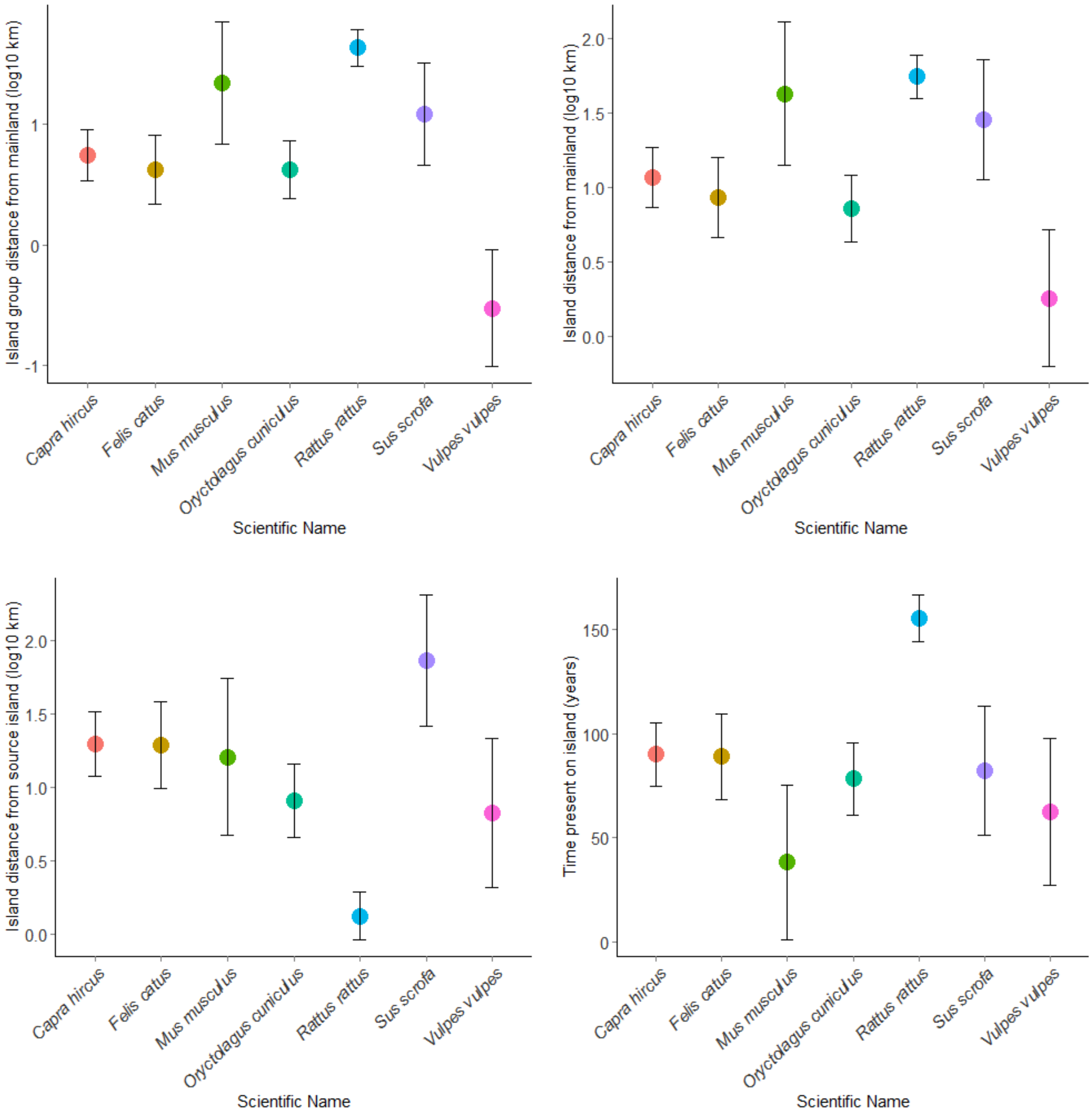

Figure 13: Differences in (i) island group distance from mainland (ii) island distance from mainland, (iii) island distance from an island hosting a source population for islands and (iv) differences in the time a species was present on islands on which eradications were attempted for each species. Points are mean model estimates and error bars are 95% confidence intervals.

3.4.3 Univariate influences on eradication success

Exploratory analysis of univariate models (Table 3) revealed that the probability of eradication success declined with increasing island size (Figure 14; GLM Coefficient: -0.4649 [0.2247 SE], $P < 0.05$), and increasing variation in ruggedness (Figure 14; GLM Coefficient: -0.6824 [0.3065 SE], $P < 0.05$). The overall effect of elevational range (Figure 14; GLM Coefficient: -0.2906 [0.2375 SE], $P = 0.22$) was weaker than the effect of variation in ruggedness, as also illustrated by the wide confidence intervals around the curve (Figure 14).

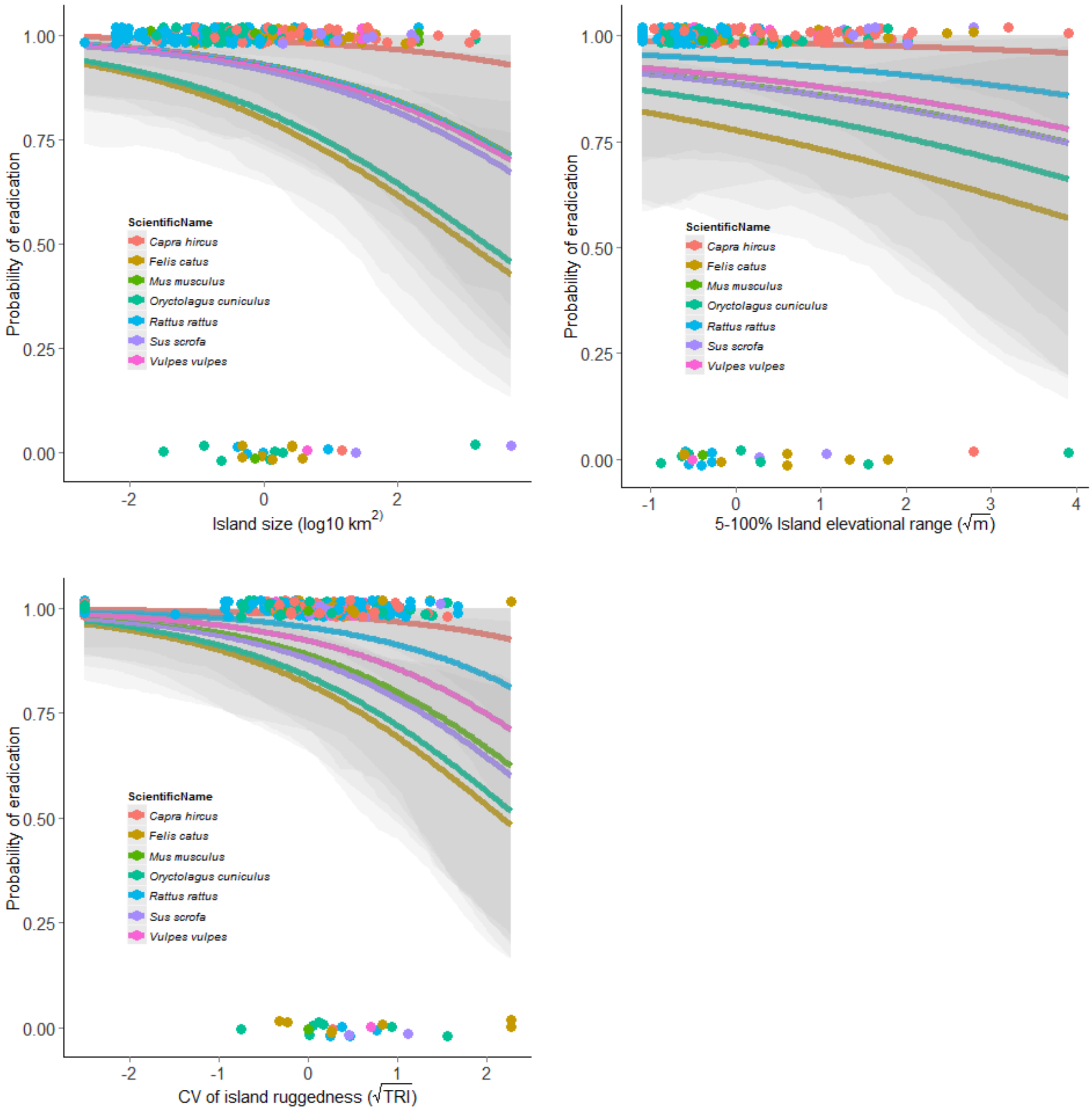

Figure 14: Univariate GLM predictions for the effect of (i) the size, (ii) the elevational range, and (iii) the variation in ruggedness, on the probability of eradication (failed = 0, success = 1). Points are alternatively jittered on the y-axis for clarity. Shaded areas are empirical 95% confidence intervals based on 200 bootstrap resamples.

The probability of successful eradication was not significantly influenced by distance of the island (or the island group) to the mainland, or the distance to the closest source island, for any of the focal pest species (Table 3; Figure 15). Similarly, the effect of the time a species had been present on the island before the eradication was attempted was not significant (GLM Coefficient: 0.0047 [0.004 SE], P = 0.24).

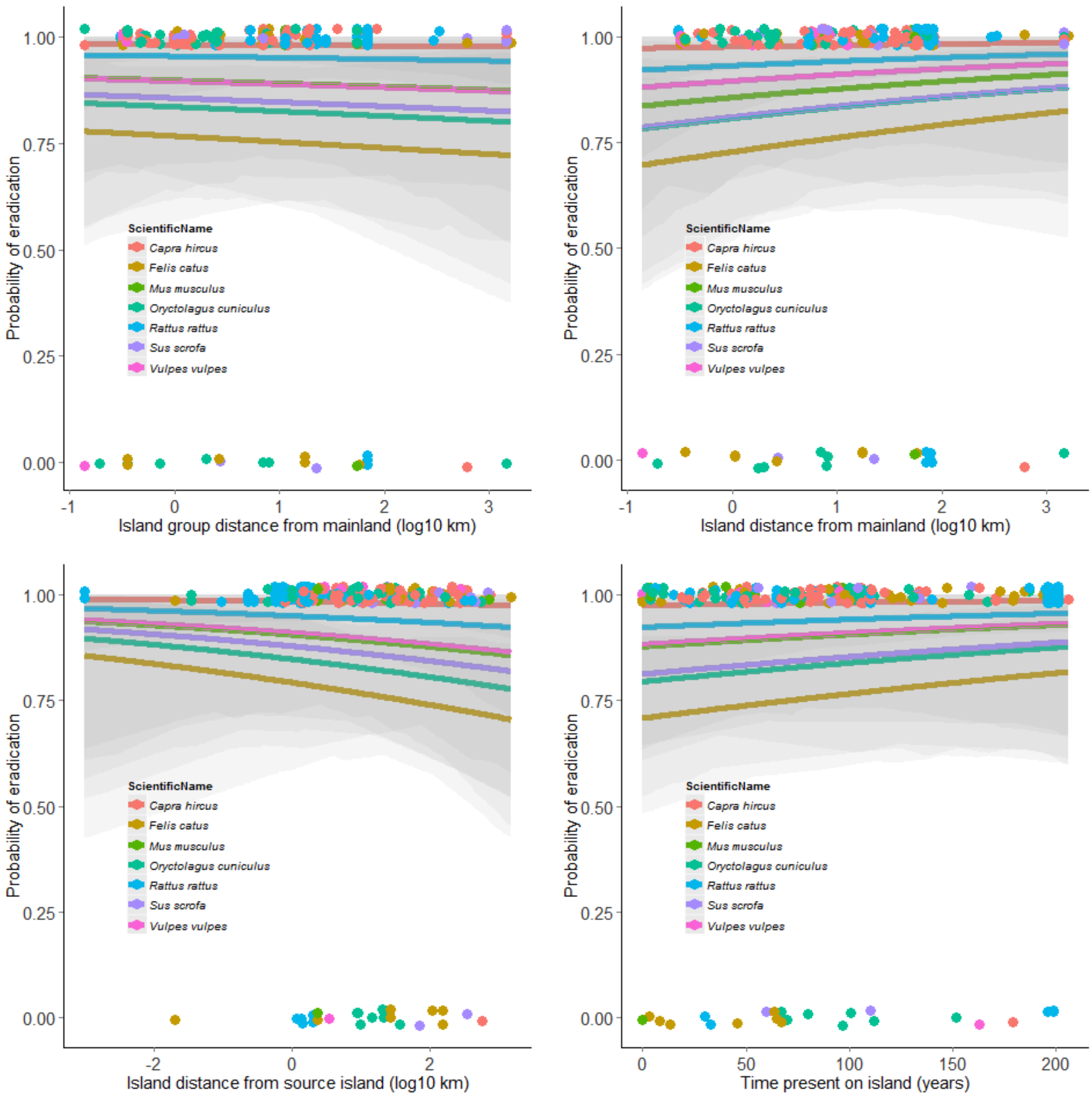

Figure 15: Univariate GLM predictions for the effect of (i) island group distance from mainland (ii) island distance from mainland, (iii) island distance from an island hosting a source population for islands, and (iv) period present on island, on the probability of eradication (failed = 0, success = 1). Points are alternately jittered on the y-axis for clarity. Shaded areas are empirical 95% confidence intervals based on 200 bootstrap resamples.

Table 3. Model estimates and test statistics for univariate GLM models for each explanatory variable. I: = intercept for individual species. Influential univariate variables are shaded in light grey.

	GLM			
Explanatory variable	Coefficient	Estimate (SE)	Z Value	P Value
Island size	I: Goat	4.249 (1.034)	4.110	0.000
	I: Cat	1.384 (0.458)	3.022	0.003
	I: Mouse	2.586 (1.128)	2.292	0.022
	I: Rabbit	1.505 (0.425)	3.541	0.000
	I: Black rat	2.572 (0.482)	5.340	0.000
	I: Pig	2.396 (0.860)	2.784	0.005
	I: Fox	2.530 (1.073)	2.358	0.018
	Variable	**-0.456 (0.223)**	**-2.046**	**0.041**
Elevational range	I: Goat	4.133 (1.032)	4.006	0.000
	I: Cat	1.249 (0.450)	2.774	0.006
	I: Mouse	2.058 (1.061)	1.939	0.053
	I: Rabbit	1.641 (0.426)	3.848	0.000
	I: Black rat	2.774 (0.482)	5.754	0.000
	I: Pig	2.050 (0.840)	2.441	0.015
	I: Fox	2.234 (1.056)	2.116	0.034
	Variable	**-0.248 (0.234)**	**-1.061**	**0.289**
Variation in ruggedness	I: Goat	4.110 (1.017)	4.039	0.000
	I: Cat	1.512 (0.487)	3.106	0.002
	I: Mouse	2.087 (1.068)	1.955	0.051
	I: Rabbit	1.646 (0.432)	3.813	0.000
	I: Black rat	3.050 (0.482)	6.326	0.000
	I: Pig	1.990 (0.788)	2.526	0.012
	I: Fox	2.481 (1.064)	2.332	0.020
	Variable	**-0.697 (0.303)**	**-2.297**	**0.022**
Distance from mainland	I: Goat	3.771 (1.044)	3.614	0.000
	I: Cat	0.990 (0.492)	2.007	0.045
	I: Mouse	1.792 (1.149)	1.559	0.119
	I: Rabbit	1.437 (0.464)	3.094	0.002

	I: Black rat	2.627 (0.656)	4.008	0.000
	I: Pig	1.457 (0.853)	1.709	0.087
	I: Fox	2.158 (1.056)	2.044	0.041
	Variable	**0.177 (0.273)**	**0.648**	**0.517**
Island group distance from mainland	I: Goat	4.009 (1.029)	3.896	0.000
	I: Cat	1.193 (0.465)	2.568	0.010
	I: Mouse	2.180 (1.113)	1.959	0.050
	I: Rabbit	1.629 (0.448)	3.635	0.000
	I: Black rat	3.057 (0.617)	4.955	0.000
	I: Pig	1.788 (0.821)	2.178	0.029
	I: Fox	2.158 (1.062)	2.032	0.042
	Variable	**-0.075 (0.250)**	**-0.299**	**0.765**
Distance from source island	I: Goat	4.149 (1.072)	3.869	0.000
	I: Cat	1.342 (0.566)	2.373	0.018
	I: Mouse	2.262 (1.112)	2.034	0.042
	I: Rabbit	1.719 (0.488)	3.526	0.000
	I: Black rat	2.957 (0.462)	6.402	0.000
	I: Pig	1.983 (0.916)	2.164	0.031
	I: Fox	2.322 (1.078)	2.153	0.031
	Variable	**-0.148 (0.262)**	**-0.564**	**0.573**
Time present on island	I: Goat	3.690 (1.064)	3.469	0.001
	I: Cat	0.888 (0.545)	1.629	0.103
	I: Mouse	1.969 (1.070)	1.839	0.066
	I: Rabbit	1.353 (0.508)	2.664	0.008
	I: Black rat	2.490 (0.726)	3.432	0.001
	I: Pig	1.467 (0.828)	1.771	0.077
	I: Fox	2.018 (1.079)	1.871	0.061
	Variable	**0.003 (0.004)**	**0.755**	**0.450**

3.4.4 An optimal model of island eradication success

Before fitting multivariate models, we inspected the correlations between explanatory variables. This was to assess (and reduce) any multi-collinearity that can inflate the importance of one variable whilst rendering another redundant. For example, the correlation between variation in ruggedness and island size (the two most influential univariate variables) was not strong (r = 0.493) suggesting that they could be assessed together in a multivariate model.

We found that variation in ruggedness was the most consistently influential explanatory variable and occurred in all five of the top-ranked models (Table 4). On its own, variation in ruggedness constituted the top-ranked model. Other variables, with less support in the top-ranked models, included island size, elevational range, time present on island, and island distance from source island.

Table 4. Comparison of the fits and ranks of the top five ranked multivariate GLM models of eradication success. The complete set of models is presented in Appendix C.

Explanatory terms	GLM Rank	AIC*c*	*k*	ΔAICc	Bootstrap % Rank 1
Variation in ruggedness	1	155.475	9	0.000	61%
Variation in ruggedness + Island size	2	156.284	10	0.808	27%
Variation in ruggedness + Elevational range	3	157.251	10	1.775	12%
Variation in ruggedness + Time present on island	4	157.300	10	1.825	1%
Variation in ruggedness + Distance from source island	5	157.434	10	1.958	0%

Overall, however, we found no substantial support for a single best-fitted optimal multivariate model. Our top five ranked candidate models were all separated by a change in AIC*c* less than 2 (Table 4), suggesting that the performance of these models was difficult to distinguish statistically (Burnham and Anderson 2002; Richards et al. 2011).

3.4.5 Investigating non-independence in eradication attempts

We performed a stratified bootstrap validation of the five top-ranked multivariate models to examine the possible influence of non-independence among ‘related’ eradication attempts. The stratified bootstrap analysis did not affect our overall finding that variation in ruggedness was the single most important explanatory variable, for the probability of eradication success, in the AusErad database. The bootstrap validation also highlighted the slight difference in performance between the 1st and 2nd top-ranked GLMs and their general improvement over the lower top-ranked GLMs (Figure 16).

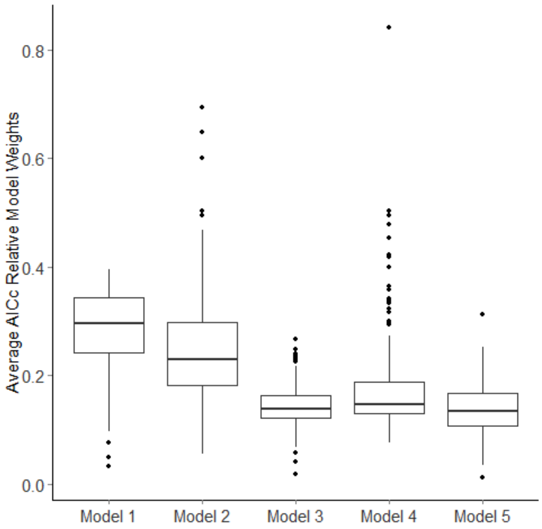

Figure 16: Boxplot showing variation in average relative model weights for the five top-ranked GLMs (see Table 4) fitted to 200 stratified bootstrap datasets with reduced possible non-independence of eradication attempts.

3.5 Discussion

Reported attempts to eradicate vertebrate pests on Australian offshore islands have been largely successful: our collation revealed a success rate greater than 92%. Although the failure rate was slightly higher (10%) when only considering the seven most frequently eradicated species (namely, black rats, feral cats, European rabbits, feral goats, house mice, red fox, and feral pigs). Our analysis of eradiaction success for these species suggests that local physical island characteristics have had the single greatest influence on the probability of eradication success. In line with our predictions, as variation in island ruggedness (a measure of the topographic complexity of the terrain) increased, the probability that the eradication would be successful decreased. A similar relationship was also observed for island area: the smaller the island, the higher the probability the eradication would be successful. Although the top-ranked model included variation in ruggedness, as the only explanatory variable of eradication success, models that also included: island area, elevational range, time present on island, and island distance from source island were difficult to statistically distinguish from this model, suggesting that these effects should not be ignored.

In the following sections, we use Australian case studies to illustrate how island ruggedness and island size, in particular, can influence vertebrate pest eradication attempts.

3.5.1 Lord Howe Island

Names	Lord Howe Island
State	Commonwealth of Australia
Location	-31.55628, 159.0885 (longitude/latitude [GDA 94])
Species	Goats, Pigs, Cats
Citations	Parkes et al. (2002) ; Hutton et al. (2007), Brown and Baker (2009), and Wilkinson and Priddel (2011)

Lord Howe Island was one of the last discovered islands in the world. Around the time of human settlement, in 1834, pigs and goats were introduced as an easily available source of meat. Black rats are thought to have established on the island after a ship wrecked in 1918, and they quickly spread across the island. Since then Lord Howe Island has witnessed the extinction of 20 endemic species: nine endemic land birds and 11 species of large flightless endemic invertebrate, largely due to predation by black rats, but certainly catalysed by habitat destruction by pigs and goats. Introduced barn owls *Tyto alba* and European rabbits were also present on the island but they are thought to have died out without human intervention.

Eradication of pigs and cats commenced in 1979 as part of the woodhen *Gallirallus sylvestris* recovery program and were successfully accomplished within the specified time frame. Vital to their success was the engagement of local hunters with local expertise and knowledge. A bounty of $15 per pig resulted in 183 being shot by island hunters by 1981, soon after which pigs were declared as eradicated. Cats were also hunted, but this was supplemented with trapping conducted by the NSW National Parks and Wildlife Service, and the island was declared cat-free by 1979. A ban on keeping domestic cats was passed in 1982 and the last de-sexed cat died in 2006.

The eradication of goats was not as straightforward. The northern part of the island had been cleared of goats by local hunters, who succeeded in removing all but a single goat, which later died. It was, however, the southern part of the island that proved more difficult, due largely to the irregular terrain (Figure 17).

Southern goats were hunted by local hunters from the ground but also by specialist hunters operating from helicopters in the air. The difficult terrain was acknowledged and a hunting strategy was drawn up that ensured that the aerial hunting drove individual goats away from the most complex topography that was inaccessible to ground hunters. In 1999, a total of 189 goats were shot from helicopters and 106 from the ground. However, in 2000, three senescent nannies were located during post-eradication monitoring.

Although the difficulty eradicating goats was proximally due to the topographic complexity of the island, together with its large size, the ultimate cause might have been diminishing funding and opposition of the local community. Conflicting value judgements by resource managers and local communities are not uncommon during eradication efforts. On Lord Howe Island, delays moving staff and hunting dogs resulted in reduced aerial hunting time; the helicopters flew an estimated 67% of the time allocated.

Eradication of rats and mice from Lord Howe Island is planned for 2015 and will involve the distribution of brodifacoum poison by hand and aerially. Introduced masked owls *Tyto novaehollandiae* are still present on the island and, given the risk of prey switching after rodent removal, an attempt will be made to eradicate them concurrently.

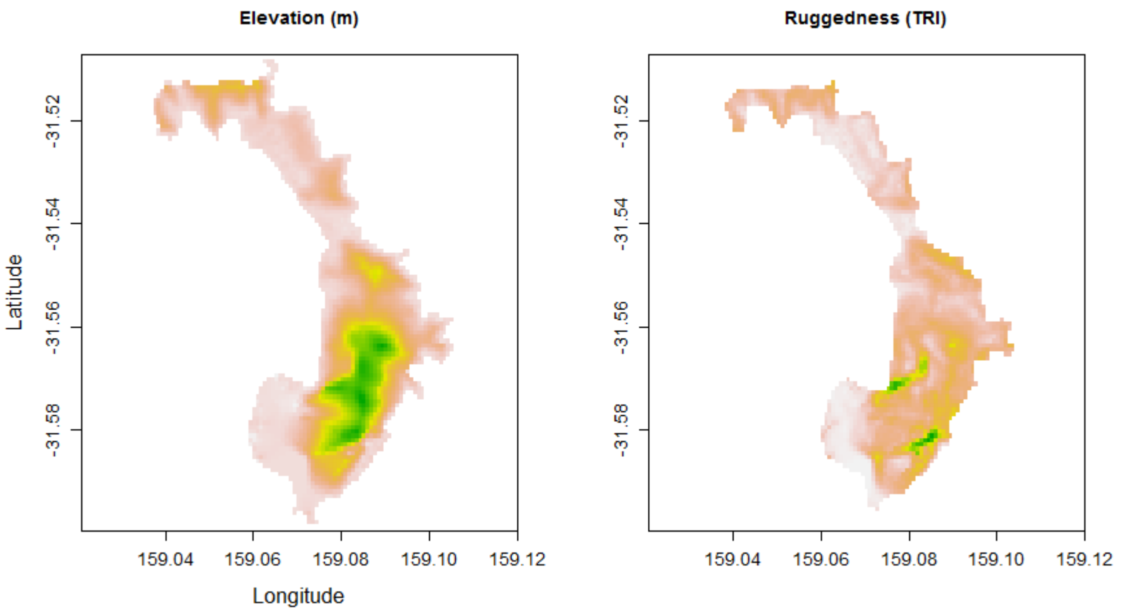

Figure 17: Maps showing the elevation and ruggedness of Lord Howe Island. The goat eradication failed because a few goats look refuge in the rugged highlands in the south of the island. On both maps, pale pink is a low value and dark green is a high value.

3.5.2 Montebello archipelago

The eradication of vertebrate pests from islands in the Montebello archipelago, during the Western Shield project, highlights how the size of an area to be eradicated can affect eradication success. The campaign was exemplary in its metapopulation approach to black rat eradication.

Names	Ah Chong, Alpha, Aster, Bloodwood, Bluebell, Brooke, Buttercup, Campbell, Carnation, Crocus, Dahlia, Daisy, Dandelion, Delta, Epsilon, Flag, Foxglove, Gannet, Gardenia, Glossypium, Hermite, Hollyhock, How, Ivy, Jonquil, Karangi, Kingcup, Lily, Middle, North West, Pansy, Primrose, Renewal, South East, Trimouille and Violet islands
State	Western Australia
Location	-20.54426, 115.69904; range: -16.38674 : -28.46433, 113.8133 : 123.0698
Species	Rats, Cats
Citations	Burbidge (2004); Howald et al. (2007); Nogales et al. (2004)

Knowledge of the Montebello Island fauna prior to 1992 revealed that four species had been extirpated following the arrival and establishment of rats and cats: the spectacled hare-wallaby *Lagorchestes conspicillatus*, golden bandicoot *Isoodon auratus*, black-and-white fairy wren *Malurus leucopterus edouardi* and the spinifexbird *Eremiornis carteri*.

In 1995 the Western Shield project *Montebello Renewal* was launched with the aim of eradicating cats and black rats from the 180 islands and islets constituting the Montebello archipelago (part of the Montebello Islands Conservation Park). It was decided to eradicate black rats before cats because rats were expected to be the primary food source for cats - it was thought that eradication would be more likely to succeed if the cats' main food source was removed.

In 1996, after a small and successful trial of black rat eradication on Renewal Island, 11,000 bait stations or plastic bags delivering 1080 (sodium fluoroacetate) baits were distributed across all 180 islands, islets and rocks by hand or from the air. Volunteers and staff monitored a range of species for non-target and secondary poisoning during 1996 but no evidence was found. Post-eradication monitoring in 1998 revealed no rats, suggesting that eradication had been successful (and no non-target or secondary poisoning effects were noted during 1997). However, rat signs were detected on the smaller islands of Alpha and Campbell in 1999 (which were immediately rebaited) and on the largest island, Hermite, suggesting that the strategy that worked for the great majority of smaller islands was not effective on the larger island. Given the past and future threat of black rat reinvasion from Hermite to surrounding islands, it was decided to rebait Hermite, and nearby Renewal, Campbell, Delta, Alpha and Bluebell islands using a helicopter-mounted bait spreader; a technique better adapted for distributing baits over large areas (Figure 18). Using New Zealand expertise and equipment, these islands were rebaited in 1999 and, following post-eradication monitoring in 2000 that revealed low-level rat activity, again in 2001. The second aerial baiting protocol proved to be more robust and no rat activity was recorded during post-eradication monitoring in 2002 and 2003.

Having eradicated black rats, cats on Hermite Island were helicopter baited with 1,100 kangaroo meat sausages laced with 1080. Post-baiting trapping, supplemented with phonic and odour lures, caught four remaining female cats and no cat activity was recorded during post-eradication monitoring in 2001, 2002 or in 2003, when eradication success was confirmed. Cats were on Bluebell Island but died out without human intervention (D. Algar, *pers. comm*). Following the removal of cats from Hermite, authorities have introduced and protected self-sustaining populations of spectacled hare-wallaby and golden bandicoot.

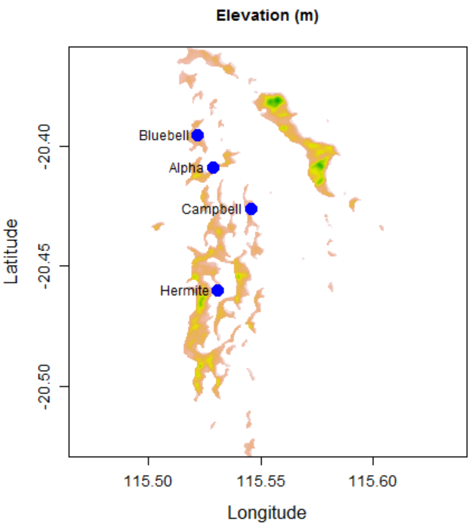

Figure 18: A map showing the size and elevation of islands in the Montebello archipelago. Pale pink is a low value and dark green is a high value.

Although we found island size to be an important determinant of the probability of eradication success, there is a growing opinion that island size (and habitat complexity) is becoming less of a barrier to the success of vertebrate eradications. This may be particularly true for larger vertebrates, which can be hunted from a helicopter (Carrion et al. 2011), or any pest species (e.g., rodents) that will take poisoned baits spread aerially from a helicopter or a fixed-wing aircraft (Clout and Russell 2006; Nugent et al. 2011). For Mexican islands, it was previously concluded that funding, rather than technical capacity or island size, are now the limiting factors preventing successful eradications of exotic vertebrate pests (Aguirre-Muñoz et al. 2008). Our analysis was conducted across historical eradication attempts for which island size might still have been an important factor in eradication success, particularly when considered relative to other islands in the same region. Although we did not have a sufficient sample to formally consider a trend in eradication success in these analyses, the relationship between mean island size on which species have been eradicated and time reveals no discernible trend; except perhaps that the larger islands have been attempted for eradication in more recent years (e.g., Towns and Broome, 2003). Many aspects of eradications are contingent on initial feasibility and planning decisions. Although not included in our models (due to the difficulty and potential bias involved in calculating it), institutional cooperation, whether financial or logistical, was also highlighted as an important factor in eradication success by several on-ground managers interviewed during the compilation of AusErad. Simberloff (2009) noted that an eradication cannot succeed even if the great majority of stakeholders cooperate in the campaign so long as a small minority allow the invader to persist on property they control. This implies that for an eradication to be successful, the existence of a person or agency with the authority to enforce cooperation is required. Cooperation in eradication efforts also includes the importance of engaging community support and public participation (Boudjelas 2009; Oppel et al. 2011). On Clarion

Island, Mexico the eradication program for rabbits failed because planning and consultation were limited and pre-eradication trials were not undertaken. The rabbit eradication campaign was stopped when previously adopted techniques were not effective enough and funding ran out (Aguirre-Muñoz et al. 2008). Several authors have highlighted the importance of consistent funding in eradication planning (e.g., Brooke et al. 2007), implementation (e.g., Donlan and Wilcox, 2008) and also as a motivation for eradication over long-term control. Parkes et al. (2002) calculated that it would be cheaper to eradicate goats from Lord Howe Island than to control their numbers for 20 years, assuming no inflation or contingency costs.

The importance of sufficient and consistent funding is well-illustrated by the example of the successful cat eradication on Tasman and Wedge islands, Tasmania. Domestic cats were introduced to Tasman and Wedge islands in 1913 and 1939, respectively, and have been significant predators of seabird populations on both islands. It was estimated that approximately 50 cats were killing 50,000 sea birds on Tasman Island per year (Saunders 2008). Following this revelation, the decision was taken to eradicate cats from both islands. However, early attempts failed due to insufficient institutional cooperation and funding. On Wedge, the first attempt in 2003 had no budget and no on-going support. Institutional cooperation was subsequently agreed for follow-up attempts in 2003 and 2004 and these appear to have been successful. Similarly on Tasman, an early eradication attempt failed due to the lack of a dedicated budget but succeeded when a second attempt was supported by a donation from a local tour operator, enabling the work to be conducted in a feasible and systematic manner (Sue Robinson and Luke Gadd, *pers. comm.*).

Although AusErad includes other examples of eradication failures, due to insufficient or inconsistent funding, these data proved extremely difficult to obtain. The sources and amount of funding are rarely adequately recorded (Donlan and Wilcox 2008): the cost of eradication attempts was recorded for fewer than 10 cases in our database, and these rarely included a comprehensive breakdown of costs. A challenge to future analyses of eradication attempts will be to obtain an unbiased variable to represent institutional cooperation, perhaps including a measure of the cost and consistency of funding for each eradication attempt.

4 Mainland eradication attempts

If a vertebrate pest species inhabits spatially-isolated habitat ‘islands’, then eradicating it from such localities across mainland Australia could theoretically be feasible using the same framework implemented to successfully eradicate it from groups of offshore islands (Parkes 1993). In practice, however, continent-wide eradication of a well-established vertebrate pest species has never been achieved (Bomford and O'Brien 1995). The principle reason for this is that if the species can reinvade habitat islands, before it is eradicated from all other occupied habitat islands (Parkes 1993), this reinvasion is in violation of Criterion 2 of the guidelines for successful eradication (Section 1.4.2).

Some of the most ambitious attempts to remove vertebrate pests worldwide have been undertaken in Australia, using poisoning and hunting, in combination with mechanical suppression of immigration. The need to control vertebrate pest species was recognised soon after European settlement and is marked in history by large-scale endeavors such as the ‘dingo fence’ and the ‘rabbit-proof fence’ (Section 1.4.1) It is likely that these programs were designed to be remedial measures to facilitate the long-term control of the populations, rather than eradication measures. Other, smaller programs have, however, successfully used fencing to enable local eradication by suppressing immigration.

Fences are used to suppress immigration of terrestrial vertebrate pests into a contained area, although often at considerable expense (Somers and Hayward 2012). Lake shores and river banks have the same effect for freshwater vertebrate pests, confining them to an isolated area that is susceptible to eradication, but in contrast to fences, they are natural barriers that incur minimal expense. Removing the cost of suppressing immigration, often the largest cost in a terrestrial eradication attempt, means that localised freshwater eradication attempts are comparatively inexpensive. Consequently, eradication attempts in contained freshwater bodies are common, often being conducted by private landowners. Ayres and Clunie (2010) recently reviewed the literature on Australian eradications of invasive freshwater fish.

In the following sections, we provide an overview of eradication attempts on mainland Australia. Since there have been no recorded continental-wide eradication attempts, we then review eradication attempts from fenced areas and freshwater bodies. There is one exceptional circumstance in which it could be argued that a species has been eradicated from a continent: when the species is removed before it is widespread or established throughout its potential range. Rapid responses to recent invasive vertebrate species ‘at large’ (i.e., that are detected but not thought to have established) have been successful (Henderson et al. 2011). For example, at least 14 mammal, 11 bird, nine reptile and three amphibian species not previously recorded in Australia were detected between 1999-2010, and the majority of these were single individuals or pairs that were extirpated shortly after detection (Henderson and Bomford 2011; Henderson et al. 2011).

4.1 Summary of mainland eradication success

There have been at least 296 eradication attempts from 162 locations on the Australian mainland (Figure 19). Most eradication attempts were conducted in Tasmania (n = 61), the Northern Territory (n = 48) and Western Australia (n = 46). Similar numbers (27 – 34) were attempted in New South Wales, Queensland, Victoria and South Australia. The fewest attempts were recorded for the Australian Capital Territory (n = 22). It should be noted, however, that the number of eradication attempts is a minimum estimate because eradication attempts spanning several unspecified locations were often reported as “various locations”.

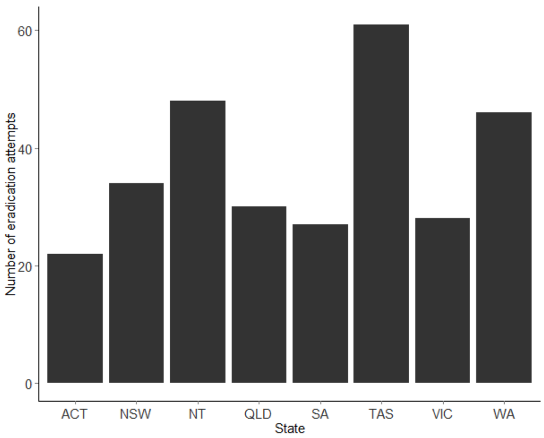

Figure 19: Histogram of the distribution of mainland eradication attempts across Australian States and Territories, including Tasmania.

The outcome of 37 recorded eradication attempts was unknown. Of the remaining 234 attempts, 193 (82.5%) were successful and 41 (17.5%) failed (Figure 20). Six eradication attempts were in the planning stage, while another 19 were in the post-monitoring stage.

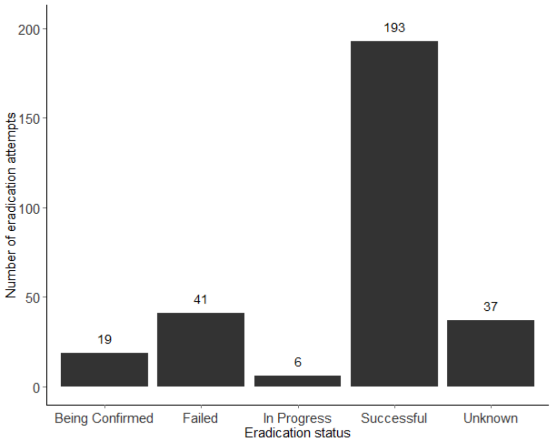

Figure 20: Histogram of the numbers of mainland eradication attempt outcomes.

Among the different classes of vertebrate pests, there were more mainland eradication attempts of fish than all other taxa, followed by mammals and then birds (Figure 21). Among those eradication attempts, the proportion of failures was similar among different vertebrate classes.

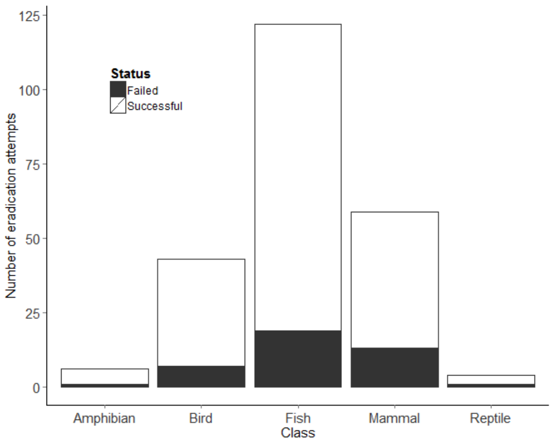

Figure 21: The numbers of failed and successful mainland eradication attempts for different classes of vertebrate pests.

Documentation of mainland eradication attempts was minimal and frequently incomplete. Species targeted and the location of the attempt were usually (but not always) recorded. Essential details, such as the outcome, the date of the eradication attempt, and the method used were rarely documented. Subsequently, we did not pursue a robust formal statistical analysis of mainland eradication attempts. Rather, we collated the AusErad records in search of commonalities to provide specific insights and generalisations of mainland eradication attempts.

4.2 Eradications on land using fences

Worldwide, fences have been used to control the movements of species ranging from African elephants *Loxodonta Africana* in South Africa to European rabbits in Australia. Fences are used for a variety of purposes, such as protecting agriculture or preventing transmission of disease, and have become a common tool for protecting threatened flora and fauna by excluding vertebrate predator and herbivore pests – such as foxes and rabbits. Following fencing, it is possible to eradicate unwanted vertebrate pest species from within the enclosure. For example, Young et al. (2013) reported on the eradication of multiple invasive predators from a 20-ha fenced seabird breeding site on the most densely inhabited Hawaiian island, Oahu. Once unwanted pest species have been eradicated, the species within the enclosure are expected to form a self-sustaining population, assuming the fence is maintained. In this way, fences have been used to protect reintroduced populations of extirpated species, such as the burrowing bettong *Bettongia lesueur* that is extinct in the wild on mainland Australia but has formed self-sustaining populations in the fenced exclosures of the Arid Recovery programme in South Australia (Read et al. 2011).

Fences can be simple mechanical barriers, but can also be equipped with additional deterrents, such as an electric charge. They must be designed carefully to minimise the probability that they are breached by the species being excluded, taking into account their ability to dig, climb and jump. Unusual events, such as high winds or spring tides, which might render them ineffective, must also be considered. There are a variety of fence designs implemented to overcome some of these issues. For example, a ‘floppy-topped’ fence is designed to prevent vertebrate pests climbing over it by introducing an unstable overhang, and is often used in conjunction with electrified wires.

Fencing, as a tool for wildlife conservation in Australia, has been reviewed by Long and Robley (2004), with a focus on fence design. We supplemented their review with a number of omitted fenced reserves. We searched the websites of the State departments responsible for managing parks (Table 5) and private conservation organisations (e.g., Australian Wildlife Conservancy) using Google and a variety of search terms that included the phrase “fence site”. By using Google, we were able to search documents (i.e., .pdf, .doc and .docx files) as well as websites. We then reviewed available information on all fenced reserves and excluded those in which a vertebrate pest population eradication had not been attempted.

Table 5. State department websites (at the time of searching) from which examples of fenced reserves were obtained.

State	Department	Website
ACT	Territory and Municipal Services	tams.act.gov.au
NSW	Environment and Heritage	environment.nsw.gov.au
NT	Parks and Wildlife Commission NT	parksandwildlife.nt.gov.au
Qld	Department of National Parks, Recreation, Sport and Racing	nprsr.qld.gov.au
SA	Department of Environment, Water and Natural Resources	environment.sa.gov.au
TAS	Parks and Wildlife Service	parks.tas.gov.au
VIC	Department of Environment and Primary Industries	dse.vic.gov.au
WA	Department of Environment and Conservation	dec.wa.gov.au

In addition to those highlighted in Long and Robley (2004), our search revealed a further eight Australian reserves fenced to exclude vertebrate pests. Removing those in which eradications have not been attempted reduced the total number to 23. According to our search, the Northern Territory and Western Australia have the largest number of fenced reserves at six, and Queensland has only a single fenced reserve (Figure 22). Interestingly, we did not locate any fenced reserves in Tasmania which, assuming our search criteria were sufficient to locate details of fenced reserves, could suggest that the former absence of red foxes obviated the need for fences.

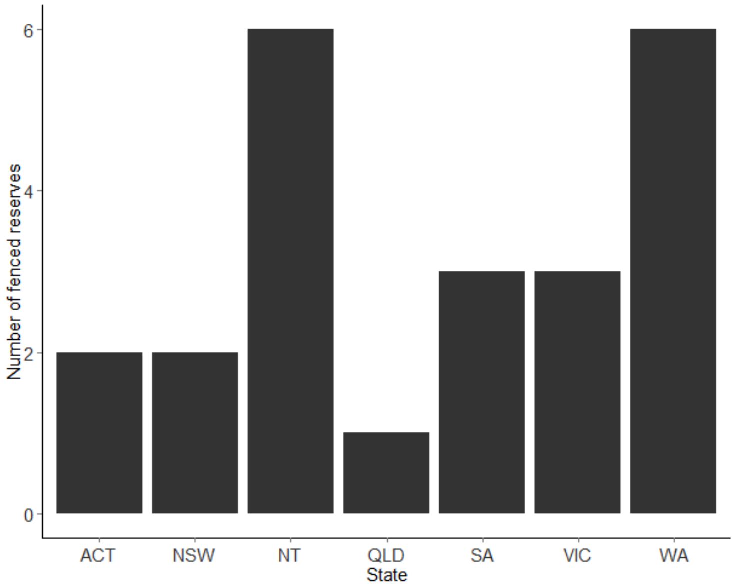

Figure 22: The number of mainland fenced reserves by Australian State and Territory, none were found (or reported) for Tasmania.

Of the fenced reserves in which a vertebrate pest eradication was attempted, it appears that the median (and mean) reserve size was largest in South Australia and was smallest in Victoria and Northern Territory (Figure 23).

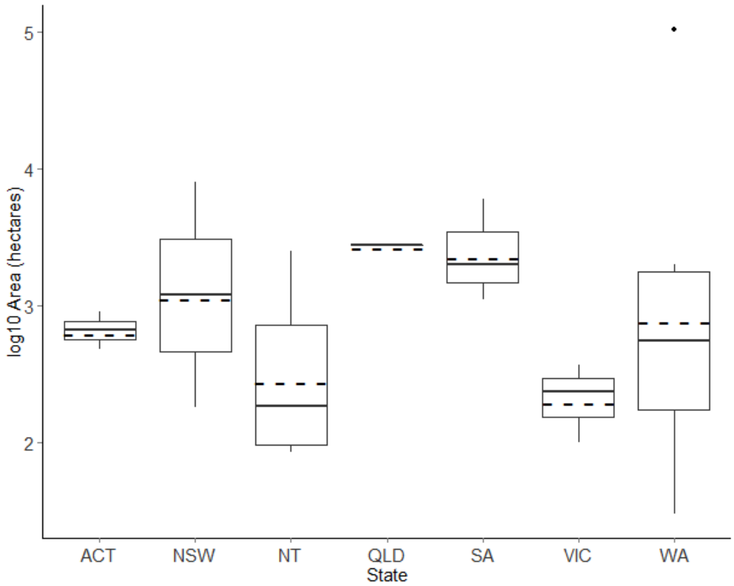

Figure 23: Boxplots of the average (dashed line) and median (solid line) fenced reserve area by State.

With only a few exceptions, eradication attempts in fenced reserves have been reported as successful. This is perhaps unsurprising given that: (i) the fences were often erected to enclose relatively small areas; and (ii) enclosures were often erected to exclude large and conspicuous species, such as kangaroos, ungulates, and invasive predators, which could be removed relatively easily in confined areas. For example, the Mac Clark and Henbury Meteorites conservation reserves were fenced to prevent overgrazing of Waddy wood *Acacia peuce*, a threatened Acacia species, and erosion of meteorite impact craters, respectively. Large ungulates (camels and cattle) inside these fenced reserves were almost entirely eradicated by ground hunting.

The only unsuccessful eradication attempts recorded in fenced reserves were of foxes and cats from the Peron Peninsula (WA); foxes, cats and rabbits from the Tidbinbilla enclosure (ACT); goats, cats and rabbits from the Yookamurra Sanctuary (SA); and cats from Venus Bay Conservation Park (SA); i.e., four of 23 fenced reserves (or 82.6% success).

4.2.1 Peron Peninsula – Project Eden

Names	Peron Peninsula
State	Western Australia
Species	Foxes and Cats
Citations	Morris et al. (2004); de Tores and Marlow (2012)

Project Eden was instigated under the auspices of the Western Shield project of the Western Australia Conservation and Land Management department in 1995. It had the ambitious aim to "*translocate and reconstruct the pre-European fauna*" of the Peron Peninsula that had been invaded by foxes and cats (among other vertebrate pest species). To achieve this aim, it was decided to isolate the peninsula from the mainland at the 3.4 km wide isthmus using an electrified fox- and cat-proof fence and then control – with the intention to eradicate - their populations on the peninsula.

The project was anticipated to proceed in three stages: introduced predator (fox and cat) control, native species reintroduction and nature-based tourism. Although not explicitly set out as an objective of the project, fox and cat eradication was the ultimate aim of the project.

Methods used to control fox and cat populations on the peninsula were the same as those used on islands, including aerial baiting with 1080 poisoned baits. The vertebrate pest populations responded dramatically. Approximately 95% of the fox population has been eradicated. Despite not reaching the targeted fox and cat densities in the allotted time frame, the decision was taken to reintroduce a number of native species populations onto the peninsula. Post-release monitoring of the populations indicated that cat predation was a major source of mortality, even though the cat density was thought to have been reduced through control.

If eradication was the ultimate objective of the Project Eden fox and cat control programme, then the attempt would be recorded as unsuccessful, as only 95% of the foxes and between 50-80% of the cats were removed. More worrying, however, is the observation that the reduction in fox numbers precipitated an increase in the cat population (a phenomenon called mesopredator release) that then predated on the recently translocated populations of native species (a so-called surprise effect; see Section 5.1.4).

4.2.2 Mulligans Flat – Gooorooyarroo Woodland Experiment

Names	Mulligans Flat – Goorooyarroo reserves
State	Australia Captial Territory
Species	Eastern grey kangaroo, foxes, cats
Citations	Shorthouse et al. (2012); Manning et al. (2011)

The Mulligans Flat – Goorooyarroo Woodland Experiment was established in 2004 as a collaborative project between the ACT Parks service and the Australia National University. The overarching goal was to restore an important area of critically endangered grassy box-gum woodland, which had been degraded by *c*. 150 years of cattle and rabbit grazing, and more recent overgrazing by the native Eastern grey kangaroo *Macropus giganteus*.

A key component of the project was the use of an exclusion fence. The 11.5 km long fence cost an estimated $1.3m and encloses approximately 485 hectares (or 63.4%) of the Blakely's red gum *Eucalyptus blakelyi* and yellow box *E. melliodora* habitat in the Mulligans Flat reserve. The enclosure is known as the Mulligans Flat Woodland Sanctuary and has facilitated the removal of pest vertebrate predators, including foxes and cats. The enclosed rabbit population has also been targeted and has been dramatically reduced. Reintroductions of Eastern bettong *Bettongia gaimardi* and brown treecreepers *Climacteris picumnus* have already occurred, and between Autumn and late-October 2013 up to 100 New Holland mice *Pseudomys novaehollandiae* were released.

In addition to facilitating the eradication of foxes and cats from the Sanctuary, the enclosure has also restricted the numbers of kangaroos. This is expected to ameliorate over-grazing pressure that destroys important habitat for re-introduced native animals, as well as promoting recruitment of vegetation.

The Mulligans Flat – Goorooyarroo Woodland Experiment is an example of the increasing cooperation between park authorities and land managers with universities and researchers to design, manage and monitor species reintroduction and preservation attempts. Ongoing public participation with such a project is paramount (Boudjelas 2009): for instance, Mulligans Flat Woodland Sanctuary contains many gates that must be kept closed with the public's cooperation.

Another well-established example of this type of collaboration is the Arid Recovery project in northern South Australia. This program has met considerable success, in part due to its collaboration with scientists from the University of Adelaide. However, inconsistent funding still managed to hamper this collaborative eradication attempt, as was seen between 2004 and 2009.

4.2.3 Arid Recovery

Names	Arid Recovery
State	South Australia
Species	European Rabbits
Citations	Read et al. (2011)

After a crash in rabbit numbers following a rabbit haemorrhagic disease outbreak in 1996, the Arid Recovery project was established with the aim to reintroduce formerly extirpated native species to vertebrate pest free enclosures in northern South Australia. Between 1997 and 2001, intensive rabbit control was conducted in four paddocks covering 60 km^2 and a total estimated 4,000 rabbits were removed by a range of methods. Successful eradication was achieved in separate paddocks by poisoning with 1080, warren ripping and burrow fumigation, trapping, and then tracking and shooting of individual rabbits. At 30 km^2, the Northern Expansion paddock is the largest enclosure from which rabbits were successfully eradicated (in 2000-2001), after techniques had been honed in three previous smaller paddocks.

In 2004, an additional paddock was added to the Arid Recovery project and was earmarked for rabbit eradication. Between 2004 and 2009 another 4,000 rabbits were removed using the same combination of methods as before, and yet the rabbits were not eradicated. It seems that resources, and particularly their application, were inconsistent, and so the program stalled and was ongoing for six years. Essentially, this provided a period when the birth rate was exceeding the removal rate, allowing the maintenance of the population.

4.3 Eradications from freshwater bodies

Ayres and Clunie (2010) reviewed the diffuse published and verbal records of pest fish eradications in Australia, of which 169 records were incorporated into AusErad. Briefly, they found that eradication attempts were rarely documented and those that were missed vital information from which they could evaluate, or learn from, a program’s success. The majority of cases recorded a successful eradication outcome, although it is not clear whether this was due to reporting bias. Unsuccessful eradication attempts were poorly documented and, in most cases, no underlying cause of failure could be assigned.

It appears that over half the eradication attempts were conducted post-2000, perhaps suggesting a change in policy, attitudes or awareness. Fish eradication has been attempted in all States and Territories, although the large uncertainty in the number of attempts precludes their comparison. The species targeted were principally common carp *Cyprinus carpio*, rainbow trout *Oncorhynchus mykiss*, Mozambique tilapia *Oreochromis mossambicus* and Eastern gambusia *Gambusia holbrooki*. Of these, it seems that common carp were the most frequently targeted for eradication, and often in closed water systems on private lands, particularly farm dams.

Methods used to eradicate fish can be generalised into two types: chemical and mechanical. Chemical eradication was more common, using poisons such as rotenone and calcium hydroxide (Figure 24). Mechanical eradication was less common and involved techniques such as draining, electro-fishing and explosives. These methods were much less effective than chemical methods: assuming the same biases apply to reporting of eradication success for chemical and mechanical methods, then the former was almost six times more effective than the latter (ratio [Success/Failure]: chemical = 8.7, mechanical = 1.5).

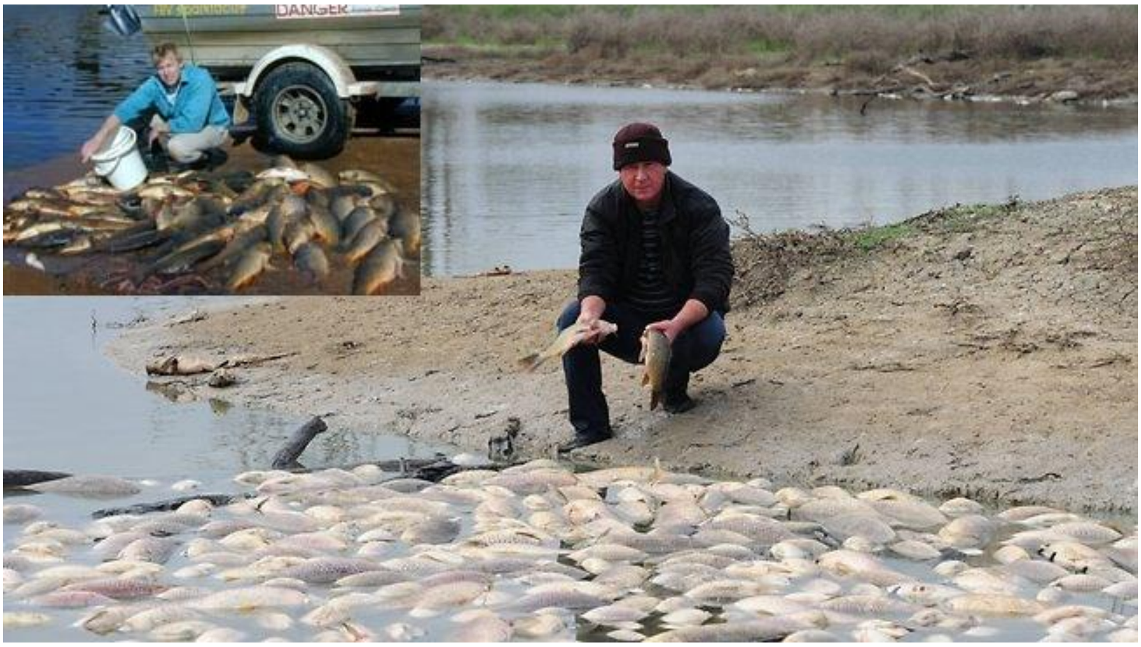

Figure 24: Shane Warwick pictured with dead European carp in a backwater of the River Murray (2011), and (inset) Brad Tucker displaying carp captured during a project to identify carp breeding hotspots. Photo source: http://resources3.news.com.au/images/2011/09/10/1226133/829319-european-carp.jpg; www.dpi.nsw.gov.au.

4.4 Other mainland eradications

Apart from eradication attempts of vertebrate pest populations within fenced reserves and isolated freshwater bodies, the remaining eradication attempts in AusErad are idiosyncratic and poorly documented. Records are from all States and Territories and most of these documented a species being hunted or trapped to local extirpation; often a recent national incursion such as an escaped pet or a State incursion of an already established species (Henderson et al. 2011). There are representatives from all the major vertebrate taxonomic groups: amphibians, reptiles, birds and mammals (fish are reviewed above), and there are almost as many species as eradication attempts, ranging from red-eared slider turtles *Trachemys scripta elegans* to blackbuck antelope *Antilope cervicapra*.

Depending upon when a species should be deemed as “established”, there are at least two examples of the successful eradication of a terrestrial vertebrate pest species from mainland Australia: American grey squirrels *Sciurus carolinensis* were eradicated from Ballarat in Victoria (Seebeck 1984), and Adelaide city in South Australia (Peacock 2009).

4.4.1 Grey squirrels in Adelaide city

Names	Adelaide city
State	South Australia
Species	Grey squirrels
Citations	Peacock (2009); Seebeck (1984)

Whether introduced deliberately by Adelaide residents transporting specimens from Melbourne or Ballarat, Victoria, or escapees from Adelaide zoo, grey squirrels were present in metropolitan Adelaide between 1917 and 1922. Anecdotal accounts suggest that they occupied an area of at least 40 km^2 where European trees were growing and residents provided food, although it is not clear how abundant they were.

It appears that the fast reaction to the establishment of the squirrel population was vital to their successful eradication. This was perhaps driven by the knowledge that other introduced species (namely the red fox and European rabbit) had become widespread socio-economic pest species. The City's government took decisive action, employing staff to hunt the squirrels. In addition, it seems that bounties were paid for the squirrel's tails, although it is not clear from where the money was provided.

It seems likely that the squirrels were somewhat restricted in their distribution, perhaps to areas of high densities of European trees. This restricted distribution most likely contributed to the successful eradication campaign. There is also some suggestion that the Australian native habitat was not suitable for them, whether due to a lack of suitable resources or overwhelming competitors. This is supported by the observation that two other Victorian squirrel populations died out without direct human intervention, likely due to competition with brush-tailed possums, compounded by predation by feral cats.

An example of a planned nationwide eradication attempt across mainland Australia was the program to eradicate feral cattle *Bos taurus* from outside managed pastoral properties, as part of the Australian Brucellosis and Tuberculosis Eradication Campaign (BTEC). Ultimately, this eradication attempt was unsuccessful, although the eradication of brucellosis from mainland Australia was successful, and is heralded as one of most successful programs in epidemiological eradications worldwide (Pappas et al. 2006; Radunz 2006). Brucellosis is a highly contagious zoonosis caused by the bacterium *Brucella abortus* that causes pregnant cattle to abort, at considerable financial expense to livestock farmers. It is also a health concern among humans, which can be contracted from infected cattle products. Brucellosis is transmitted from animal to animal by way of direct contact and although livestock were treated for the infection, it was thought that the feral cattle population was acting as a reservoir for the disease. The BTEC was a success and Australia was declared free of brucellosis in 2002. However, the feral cattle eradication was unsuccessful, mostly due to the large number of feral cattle roaming in the Northern Territory (Lehane 1996).

Throughout this report we have considered Tasmania as part of mainland Australia due to its size and economic and social proximity to the other mainland States and Territories. In terms of geographical proximity, however, it is separated from the rest of mainland Australia by the Bass Strait; a dispersal barrier to most terrestrial and freshwater vertebrate pest species. This barrier has protected Tasmania from some of the worst exotic vertebrate pest species, including the red fox. Sadly, however, the Strait poses no barrier to humans who reportedly released several Foxes into Tasmania between 1998 and 2001, which, it is possible, have now become established. A State-wide campaign to eradicate them is currently underway.

4.4.2 Foxes in Tasmania

Names	Mainland Tasmania
State	Tasmania
Species	Red fox
Citations	Saunders et al. (2006); Sarre et al. (2012)

In contrast to the unenviable native mammal extinction record of the other mainland States and Territories, Tasmania has only recorded a single mammalian extinction – the thylacine. It is widely held that the difference (between the two landmasses) responsible for the mammalian extinctions is the presence of the red fox on the mainland. Indeed, it has been suggested that the establishment of foxes in Tasmania would place 78 species of native terrestrial vertebrates at risk of extirpation or extinction. Recent studies have used the locations of fox scats and carcasses to construct a predictive model to identify the ‘putative’ current distribution of foxes across Tasmania (Figure 25).

Figure 25: Physical evidence of foxes collected across Tasmania. Source: www.dpiw.tas.gov.au.

Fox eradication from Tasmania is complicated by the majority of factors identified in this report and other literature. These include: (1) Tasmania has a complex geology with highly variable topography and large altitudinal ranges; (2) the size of the area is large and the eradication attempt is economically expensive, and therefore prone to failure; (3) the presence of human habitation offers the foxes refuge, alternative food sources, and has meant that the eradication has suffered considerable social opposition; (4) there are a wide variety of non-target species that are at threat from an eradication attempt, and exotic vertebrate pests that can lower baiting efficiency; and (5) there is an abundance of native mammal prey, and recent massive declines in Tasmanian devil populations State-wide remove an effective competitor and a potential predator of fox kits. Alternately, since the fox is a relatively recent arrival, it might be easier to eradicate, although our island eradication results suggest that eradication can be more difficult for recent arrivals, perhaps precisely because they have only recently arrived, occur at low densities, and are still spreading.

5 Discussion

Australia is a world leader in vertebrate pest eradication attempts and has recorded at least 650 eradication attempts of 68 different vertebrate pest species since 1800. The majority of these eradication attempts have been successful, particularly when considering islands and mainland eradication attempts separately: more than 92.5% of island eradication attempts succeeded compared with c. 82.5% on the mainland. We note, however, that information on eradication attempts (and particularly mainland eradication attempts) is generally of poor quality, and patchily distributed in the literature (relying heavily on verbal accounts), and these rates should be interpreted cautiously.

From quantitative and qualitative analysis of the data collated in AusErad, it seems that certain species have been more difficult to eradicate than others. Eradication attempts of goats and black rats have generally succeeded, although there have not been any reported attempts to eradicate black rats from mainland Australia. There are a few hypotheses to explain the high eradication success of both these species. Firstly, they were often introduced in the 19th Century by whalers and early visitors, since which time they have caused extensive damage (environmental, financial, and social) and have subsequently been subject to large well-funded eradication campaigns (e.g., the Western Shield project; Section 3.5.2). Secondly, both species are usually targeted using helicopters (to hunt or distribute baits from), which are particularly efficient at covering large areas quickly and reaching otherwise inaccessible refugia. Thirdly, technological advances (such as using ‘judas’ animals and sophisticated bait delivery) have improved the success of eradication attempts for these species compared with others.

Conversely, some species appear more difficult to eradicate. Compared with goats and rats a higher proportion of cat and rabbit eradication attempts have been unsuccessful. It is likely that the individual species ecology also has a prominent role in determining these differences (Parkes and Panetta 2009). Goat and rat eradications are possibly more successful, at least in part, because of the behaviour of the former to herd (Cruz et al. 2009), and the latter to cache (toxic) baits (Taylor and Thomas 1989). Alternately, cats are solitary and being largely nocturnal and secretive they are difficult to target (Campbell et al. 2011). Rabbits have an extraordinary reproductive capacity and their successful eradication may be particularly reliant on consistent funding and campaign effort, as well as a combination of methods (i.e., a ‘wide range of techniques’; Hermes 1987) to target all individuals. On Macquarie Island, following the cessation of the *Myxoma* virus releases (2006) and the completion of aerial baiting (July 2011), the last (detected) rabbits were tracked and hunted using a combination of spotlighting, trapping, thermal imaging and dogs. The ground eradication team has covered *c*. 69,000 km in these hunting efforts, for a tally of 13 rabbits. There has not been any sign of rabbits since December 2011.

Eradication attempts of common carp were generally successful, perhaps because they were usually conducted in isolated, closed-water systems. Similarly, eradications of terrestrial vertebrate pest species from within fenced reserves were usually successful. These observations, taken with the finding that island eradications generally succeed, suggest that eradications are most likely to succeed at isolated locations where immigration is minimised. The requirement to reduce immigration to zero is a criterion in Bomford and O'Brien's (1995) guidelines for eradication success (Section 1.4.2) and scientists have been developing tools to prioritise islands for vertebrate pest eradication based on the current/future threat of immigration and reinvasion (Harris et al. 2012; Russell et al. 2009).

In the case of contained locations on mainland Australia, unsuccessful eradication was generally due to an equipment failure (e.g., a fence), or poor eradication planning (e.g., insufficient funding and lack of adequate resources). The Arid Recovery program, to eradicate rabbits from a fenced enclosure, initially failed because the fence allowed passage of young rabbits. The eradication eventually succeeded after a second, finer-diameter mesh was installed (Read et al. 2011). Failure in the eradication planning, usually due to a loss or change in funding, is also commonly cited as a reason for an unsuccessful eradication attempt, although this was difficult to quantify and is usually only recorded anecdotally.

Advances in statistical methods have facilitated the comparison of different factors (including imputation of missing data) and the prediction of outcomes of planned eradication attempts (e.g., Christensen et al. 2013; Pluess et al. 2012b). Our formal statistical analyses indicated that the topographic complexity of the eradication location is an important determinant of eradication success, at least on Australian islands and depending on the species. This relationship (Figure 15) was apparently most influential for feral cats and European rabbits, both species which are targeted 'more' by ground-based operations. To our knowledge, this has not been formally demonstrated before. Parkes et al. (2002) suggested that the goat eradication on Lord Howe Island failed because the high topographic complexity of the southern part of the island allowed animals to find refuge from ground and aerial hunters. Other unsuccessful eradication attempts could probably be attributed to topographic complexity. However, this could only be confirmed through combined meta-analysis of many eradication attempts, to disentangle this effect from other, often similar effects, such as area or elevational range. In addition, our results demonstrated that the size of the eradication area plays a key role in eradication success, at least on Australian islands. Other authors have also emphasised eradication area. For example, Pluess et al. (2012a) showed that the probability of eradication success decreased as "infestation area" increased.

Our findings are strengthened by the volume of our dataset. To our knowledge, our analysis of the 650 AusErad island records is the largest of its kind. Pluess et al. (2012b) analysed a database of 173 eradication attempts, whilst Glen et al. (2013) analysed a database of 1,224 eradication attempts. However, the latter study did not use a formal statistical analysis.

5.1 Eradication "best practice"

All eradication programs face challenges of limited resources, yet there are some aspects that are either neglected or arise as unforseen consequences, but which, we argue, should be considered wherever possible. Part of the pessimism, generally, about controlling biological invasions arises from widely publicised management failures, especially failed eradications (Simberloff 2009). By addressing knowledge gaps in eradication science and advocating "best practice" we believe that the success of eradication efforts will be increased and public perception, as well as community and government support, will be enhanced.

5.1.1 Feasibility planning

The success of eradication efforts hinges on first satisfying the criteria for eradication success (Section 1.4.2) at a feasibility stage (Parkes and Panetta 2009). We assert that there will be fewer failures if all future feasibility planning is conducted rigorously. Ultimately, however, the assessment of eradication feasibility cannot be separated from the hazard posed by the vertebrate pest, with greater resources being justified to locations experiencing the greatest threat (Brooke et al. 2007). Feasibility of eradication must therefore be viewed in the context of the minimum effort that can be achieved to satisfy the criteria and overcome the location- and species-specific constraints.

In Australia, the intergovernmental National Environmental Biosecurity Response Agreement (NEBRA) provides the decision framework, governance and responsibilities for determining appropriate actions when a vertebrate incursion is detected. A National Management Group is convened to consider, through the drafting of a national incident response plan, whether an incursion: (i) poses a nationally significant pest risk; (ii) is technically feasible to eradicate; and (iii) is cost-beneficial to do so. All three factors must be met, in addition to budgetary considerations, before a decision is taken to commence a national, cost-shared eradication program. For the technical feasibility analysis, seventeen criteria are listed in the NEBRA (Item 4 of Schedule 4), although there are clear challenges with how to quantitatively interpret these criteria.

While a 'can do' attitude is essential for successful eradication, it has also been convincingly argued that a successful eradication campaign requires meticulous planning as well as 100% commitment and effort from all members of the project team (Cromarty et al. 2002). Underlying this is the need to rigorously monitor progress so that problems can be recognised and addressed as they arise. Interestingly, Pluess et al. (2012a) found that 'reaction time' ('the time elapsing between the arrival (or detection) of the organism and the start of the eradication campaign, counted in months') and 'level of preparedness' ('none' to 'high') were unrelated to eradication success. The authors concluded that having a contingency or eradication plan is no guarantee for success. They also argued that neither should lack of knowledge about the species be an excuse for inaction (Pluess et al. 2012a). Past experience has surely taught us that we need to be bold, and take some chances with eradication, especially where the biodiversity, economic, or social benefits are very large!

Methods used in planning eradications have advanced considerably since the first eradications in the early 19th Century. Whereas locations for eradication attempts would have almost certainly been decided on a case-by-case basis in the past, there are now evermore advanced methods by which to prioritise locations for eradication attempts (Brooke et al. 2007; Harris et al. 2012; Pluess et al. 2012a). Unfortunately, we could not gather sufficient data on the financial cost, or the motivation of key individuals and organisations, to statistically include as effects of eradication success. If, however, topographic complexity (i.e., ruggedness) and island area are related to the difficulty with which to plan and conduct an eradication then they may themselves be proximate measures for financial cost (Donlan and Wilcox 2007; Martins et al. 2006). In contrast, our measures of reinvasion risk, distance of island (and island group) from mainland and potential source islands, were poorly supported, suggesting that they have been less important in determining eradication success for this sample of attempts.

In the past, most eradications (in Australia and worldwide) have been undertaken on uninhabited islands (e.g.,Towns and Broome, 2003). Human populations, and their associated pets and livestock, therefore raise issues for eradication feasibility, which were not previously (or only rarely) encountered. It is particularly important to deal with community uncertainty, which can regard both the necessity and the efficacy of eradication actions

(Blackman et al. 2013). Wilkinson and Priddel (2011) suggested that support for a pest eradication from residents of an inhabited island (or mainland area) is most likely if: (1) the threats posed by the pest are understood; (2) the eradication seems possible; and (3) the benefits that will accrue are appreciated. Support is likely to be strongest if the feasibility of the eradication can demonstrably provide benefits to the regions biodiversity and its inhabitants (Boudjelas 2009).

5.1.2 Methodological and technological advances

Just as the eradication planning methods have evolved, so too have the techniques used to implement the eradication. The first attempts to eradicate black rats from the Montebello archipelago (WA) were conducted by distributing poison baits on foot. While this was effective for the smaller islands, it failed on the larger islands, which then acted as a source population from which black rats could swim back and recolonise the smaller islands (Burbidge and Manly 2002). A second attempt to eradicate black rats, however, capitalised on the technological advances developed in New Zealand and used helicopters, equipped with Global Positioning Systems and precision bait distributors to eradicate the black rats successfully from the archipelago. The continued development of aerial baiting equipment and strategies has enabled the eradication of several rodent species, pigs and goats from a range of islands of ever-increasing size. Indeed, current practice for aerial poisoning has evolved out of over 50 years of operational trial-and-error and research, and the technique can now reliably produce kills of >95% (Morgan et al. 2006). If the ongoing aerial baiting of the brown treesnake (*Boiga irregularis*) on Guam (Clark and Savarie 2012) is ultimately successful it will be undoubtedly one of the greatest vertebrate island eradication achievements to date.

Relative to diurnal conspicuous species, cats are difficult to eradicate but Christensen et al. (2013) showed that poison bait uptake by cats is best predicted by a single variable: pre-baiting, or the deploying of bait without poison to acclimatise the animals to its presence and palatability. Prefeeding with non-toxic bait is now standard practice in the aerial poisoning of a variety of mammal pests (reviewed in Nugent et al. 2011). The prefeed is usually sown about a week before the toxic bait is sown, although two prefeeds may be more effective than one (Coleman et al. 2007). However, each additional prefeed increases the cost of the eradication attempt substantially, so Nugent et al. (2011) examined whether the extra cost produced a worthwhile increase in eradication efficacy. They concluded that prefeeding helped reduce the risk of sub-lethal poisoning not only by increasing familiarity, but also (in conjunction with high sowing rates) by increasing the bait encounter rate. Prefeeding may help increase encounter rate if it changes foraging behaviour, however, it is also important that non-toxic prefeed is not oversupplied in areas of lowest pest density (Nugent et al. 2011). Obviously, a total kill can only be achieved if every target animal has some toxic bait sown into its home range. The probability that pests accept (i.e., consume) toxic bait when they encounter it must also be very high to achieve a substantial kill. In this regard, the timing of baiting operations to coincide with periods of relatively low breeding can reduce the risk of young in the nest not eating bait (Towns and Broome 2003).

It has been speculated that island area is no longer a constraint on eradication success (Clout and Russell 2006), presumably because the effect of area can be counter-balanced by the added efficiency of using more helicopters. Although Campbell Island (NZ) is the largest island ever eradicated of rodents, sub-Antarctic South Georgia is anticipated to claim that title within a few years (www.sgisland.gs) and plans are developing in the Galapagos Islands to rival that feat (e.g., Nicholls 2013). For species that cannot be hunted from a helicopter or that are less likely to take poison bait, choice of an appropriate eradication strategy will make a significant contribution to the probability of eradication success.

The eradication of carp from Lake Crescent, in the central highlands of Tasmania was successful because it implemented an integrated combination of pest management strategies (Diggle et al. 2012). The initial incursion was contained to Lake Crescent and the upstream Lake Sorell. Carp population reduction at the lakes was achieved by direct fish down and spawning sabotage strategies that prevented recruitment. This was accomplished by deploying a combination of wire mesh and purpose-built polyethylene barrier nets to block (exclude and contain) carp access to their preferred spawning habitats in the macrophyte rich wetland areas. 'Judas' fish were surgically implanted with radio transmitters and used to identify (and locate) aggregation behaviours (and areas). Physical removal was conducted with non-targeted electrofishing (backpack and boat) and net fishing (barrier traps and gill or seine nets). Chemo-attraction trials were successful in capturing carp that had persistently evaded capture by other methods, and highlighted the vulnerability of mature breeding-driven carp. Further Steel traps and purpose-built super fyke nets were deployed along the barriers at key wetland access points to passively trap mature carp pushing into the wetlands. Around the clock deployment of these passive traps was crucial in capturing mature carp during the night and at dawn when their spawning urge is greatest. In total, the eradication caught 7,797 carp and the last wild carp was removed from Lake Crescent in December 2007. The eradication program continues in the larger, neighbouring, Lake Sorrell.

The use of 'Judas' animals (usually transmitter-bearing radio-tracked individuals) for identifying wild aggregations, habitat refugia, or low-density specific behaviours has been commonly used with herd-living ungulates (for locating hard-to-find conspecifics) but also successfully applied to birds (Woolnough et al. 2006), reptiles (Tennesen 2010), fish (Taylor et al. 2012), and rodents (Russell et al. 2005). In New Zealand, a single radio-collared Norway rat (*Rattus norvegicus*) released on to a rat-free island was not caught for more than four months, despite (supposedly) intensive efforts to trap it (Russell et al. 2005). The rat swam 400 metres across open water between islands, evading capture for 18 weeks until an aggressive combination of detection and trapping methods were deployed simultaneously.

There are many emerging technological advances that promise to improve the probability of eradication success. Integrated Geographic Information System (GIS) tools that aid eradication planning, management, and monitoring can improve the capability and campaign efficiency (Lavoie et al. 2007). Thermal imaging has been compared to visual surveys and has been shown to be more effective at locating individuals at low density (Edwards et al. 2004). The use of fencing and acoustic deterrents can improve the probability of decreasing re-invasion, and thereby increase eradication success. Alternately, fertility control, in terms of reducing a pest vertebrate population, is a promising application but is so far unproven (Parkes and Panetta 2009), and the deliberate use of pathogens as vertebrate biocontrols have been largely unsuccessful. However, the use of *Myxoma* virus and rabbit haemorrhagic disease as sequential epizootics can cause initial reduction of naïve rabbit populations allowing remaining survivors to be eradicated through baiting (Priddel et al. 2000), or alternative trapping. There have been recent advances in the progress towards species-specific poisons and deterrents for small mammals (e.g., capsaicin used to protect bird eggs Baylis et al. 2012) . Advances in the field of molecular systematics, and its application to exotic pest species management, has allowed researchers to identify invasion pathways, propagule sources, and the risks of re-invasion (Abdelkrim et al. 2007; Le Roux and Wieczorek 2009; Russell et al. 2010; Veale et al. 2012). Unfortunately, the uptake of these advances has been slow, perhaps because they are prohibitively expensive, such that the biological material required to use them has not been collected, or the technology is not sufficiently refined for widespread use.

5.1.3 Non-target species

Rarely will the species targeted for eradication be the only species present. Although black rats were successfully eradicated from Boodie Island by aerial baiting, native burrowing bettongs (the namesake of the island) were also eradicated in the same campaign and the population could only be restored by translocating individuals from another population (Morris 2002). Apart from species at direct threat from the eradication method, those planning eradication attempts should also consider species that might be susceptible to indirect effects. For example, when planning the eradication of black rats from Pinzon Island in the Galapagos archipelago, managers took the decision to cage a large sample of Galapagos hawks *Buteo galapagoensis* to ensure that at least some of the island population would be prevented from eating carcases containing brodifacoum poison.

Preventative measures for non-target lethality can usually be foreseen. During the eradication of the house mouse from Thevenard Island (WA), a tailored poison-delivery station was developed in which the small entrance hole could be used only by the smaller exotic species and not the larger native short-tailed mouse *Leggadina lakedownensis*. By using this station type, and carefully timing baiting to avoid the period when young (small) native mice would be present, this eradication successfully targeted the pest species (Moro 2001).

Pre-eradication assessments of bait palatability, uptake rates, application methods, and movement of toxins within the ecosystem are commonplace when planning a successful eradication effort, and evaluating the impacts of poison bait on non-target species (Clout and Russell 2006; Courchamp et al. 2003; Rodríguez et al. 2006). In particular, biomarker studies employ non-toxic pellets containing a dye that fluoresces under ultraviolet (UV) light. The amount and location of the fluorescence can be appraised to address questions (pre-eradication) about uptake frequency, animal movement, and dosage levels (Savarie et al. 1992).

In some cases, bait interference can dramatically lower the efficacy of the eradication program itself. On tropical islands, a particular problem is the consumption and removal of baits by land crabs (Cuthbert et al. 2012). Land crabs do not experience the toxic consequences of brodifacoum (Pain et al. 2000). When they consume bait pellets, land crabs are a potential impediment to a successful eradication, simply because any pellet eaten by a crab is not available to a rat. On Henderson Island, in the South Pacific, Cuthbert et al. (2012) estimated hermit-crab densities in areas of high abundance, assessed crab bait consumption rates, and determined the required baiting application rate to ensure that 100% of Pacific rats (*Rattus exulans*) consumed bait in areas with high crab densities. This is similar to quantitatively determining the effect of sowing rate and sowing pattern on the effectiveness of bait exposure (Nugent et al. 2011).

While post-eradication environmental sampling (and subsequent analysis) adds labour and operating cost to eradication programs, monitoring data from completed eradications have undoubted value in supporting future risk assessments. This is particularly important for addressing information gaps around eradication practices and gaining support from the media as well as appeasing public concern (Fisher et al. 2011).

5.1.4 Surprise effects

Related to the indirect effects of eradication attempts are unexpected effects, also known as *surprise effects*. In contrast to mitigating secondary poisoning to non-target species, these effects are unforeseen consequences attributable to an eradication attempt and usually involve changes in an untargeted population. Introduced exotic pest species, and perturbed native pest species, play functional roles in the ecosystem and are subject to the same ecological processes as more natural systems (Shea and Chesson 2002). It is not unexpected, therefore, that their removal will precipitate a variety of ecosystem change, not all of which will benefit every species. Several eradications of exotic herbivores have been linked to dramatic increases in exotic plant populations. A notorious example of this is the population explosion of the exotic vine *Operculina ventricosa* on Sarigan Island, following the removal of feral pigs and goats, which until then had held them at very low density (Kessler 2002).

The best-known theoretical case-study for surprise effects is the unanticipated increase in bird predation as a consequence of removing rabbits, whereby cats switch to preying on birds when the rabbit population density is dramatically reduced (Courchamp et al. 1999; Courchamp et al. 2000). Other examples include ecological changes called *mesopredator release* and *hyperpredation*, whereby the removal of one predator is overcompensated by an increase in another (perhaps more efficient) predator (Hanna and Cardillo 2013).

In most cases, the ecological consequences, of removing a vertebrate pest species from a particular location, have not been well recorded. These consequences may include longer-term changes that are not easily quantified or changes in under-explored aspects of the system, such as in the soils and invertebrate communities (Crooks 2002; Ehrenfeld 2010). It is tempting to believe that eradication of vertebrate pests (particularly browsers and predators) will halt further degradation of a native biotic community; because normal restorative processes, of plant and animal succession, are sufficient to reverse the trend (Atkinson 2001). However, Mulder et al. (2009) found that the successful eradication of rats on islands is by itself unlikely to result in restoration to an uninvaded state. They found that most impacts of introduced rats were mediated through seabird density, but concluded that even if seabirds recolonised after eradication an entirely novel plant community is likely to emerge (Mulder et al. 2009).

Together with the known direct and indirect, as well as the anticipated and unanticipated, potential consequences of an eradication attempt, it is clear that eradication attempts should not just monitor the target species but also other non-targeted species. Such groups of species could include those that are particularly indicative of healthy ecosystem functioning, e.g., pollinators (Anderson et al. 2011). This is a research area that is under-developed in eradication planning and monitoring, and will greatly benefit from future research.

5.1.5 Pre- and post-eradication monitoring

Monitoring both target and non-target species before and after the eradication attempt is essential. Pre-eradication monitoring data is required to usefully prioritise the need for an eradication attempt at a particular location relative to other potential locations. This is all the more pressing when funding to conduct eradications is limited. Some pre-eradication data are usually collected, for it is only through these data that we know that there is a pest species that needs to be eradicated. While collecting preliminary data, however, it is relatively straightforward to collect data on additional aspects of the ecosystem that might be vulnerable, or indicative of cascading changes, due to a future eradication attempt. Often, these data inform us about how the eradication should proceed and, perhaps most importantly, when it should stop. Ramsey and Will (2012) propose using pre-eradication monitoring data to calculate the detection probability of the target species, facilitating how the eradication progress is monitored and when the eradication attempt can be stopped.

One of the most controversial eradication campaigns currently underway in Australia is the eradication of foxes in Tasmania (see also Section 4.2.2). Genetic analysis collected in Tasmania has identified at least 18 individual foxes, including both males and females (Sarre et al. 2012), and physical evidence collected since 1998 has included carcasses (4), one skull, two sets of footprints, one blood sample and 61 DNA-positive scats. Since 2000, the total cost of the fox eradication effort (including funding for the Fox Free Taskforce (2001-2006)) has been over AUD$35 million. However, there is a conspicuous lack of verifiable sighting data and there has been no fox activity collected since July 2011; the last road-kill carcass was discovered in 2006.

Post-eradication monitoring can be intensive and costly but it is also essential for confirming the success of the program. If post-eradication monitoring reveals that an eradication attempt has failed, then a follow-up attempt will be most likely to succeed if it is conducted quickly, before the population has rebounded to levels observed pre-eradication attempt. Post-eradication monitoring of the Western Shield *Montebello Renewal* project revealed that the first attempt had failed to eradicate black rats from the archipelago, allowing the managers to return quickly for a follow-up attempt that was subsequently successful (Morris 2002). Alternately, however, failure to detect more animals at the apparent end of an eradication does not necessarily mean that no more individuals are remaining (Morrison et al. 2007). Morrison et al. (2007) described a conceptual approach to increase the likelihood that any inability to detect additional 'Lazarus' animals indicates successful eradication, rather than the individuals having become better at escaping detection. In addition, a growing literature provides methods for robust decision making when declaring an eradication successful (Panetta et al. 2011; Rout et al. 2013; Rout et al. 2009a; Rout et al. 2009b; Solow et al. 2008). These tools allow managers to calculate the optimal solution for declaring successful eradication, over a range of possible costs of continued management and surveillance, and allowing for the uncertainty in estimating the consequences of declaring eradication prematurely.

Equally important, to knowing whether the target pest has been eradicated, is knowing whether the ecosystem, or a non-target species, has been unduly perturbed by an anticipated or unanticipated secondary effect. A comparison between pre- and post-eradication monitoring states can highlight effects that the eradication had on the system using a Before-and-After-Control-Impact (BACI) assessment, for which mature statistical analyses have been developed (Courchamp et al. 2003). It is essential to be able to demonstrate the changes (and benefits when they occur) to the ecosystem after eradication efforts in order to assist with leveraging funds for future operations.

5.1.6 Welfare and ethical awareness

Historically, eradication trials from uninhabited islands have not generated the same ethical controversy as mainland control operations (Cowan and Warburton 2011). The majority of island locations in AusErad were uninhabited by humans. However, the need to integrate animal welfare practices for pest animals in any eradication program has become increasingly recognised (Braysher 1993; Humane Vertebrate Pest Control Working Group 2004; Oppel et al. 2011).

Ethical issues pertaining to eradications are substantially complex because they relate to both the concept of eradication and to the method undertaken (Cowan and Warburton 2011). The '*Australian Code for the Care and Use of Animals for Scientific Purposes*' (the Code) provides an ethical framework and governing principles to guide the decisions of pest managers in eradication programs. While the Code states that investigators must avoid using death as an experimental end-point whenever possible (Item 1.27), eradication programs, by definition, almost always involve killing; often of large numbers of animals. Eradication programs are therefore required to provide and justify the planned end-point (Item 2.2.16). While eradication methods may be deemed 'humane', pest managers still face the ethical challenge to justify the number of animals killed in relation to the conservation benefits achieved.

Eradication methods have welfare issues for both target and non-target animals. For instance, 70% of rodent eradication attempts on islands use non-selective toxins, which can result in secondary effects such as non-target poisoning and bioaccumulation (Howald et al. 2007). Different control tools have different welfare impacts, emphasising research into the relative humaneness of control methods. In Australia, eradication programs must adhere to species-specific humane codes (www.feral.org.au/animal-welfare/humane-codes/). These codes are accessible to the wider community to minimise animal suffering during eradication and control actions conducted not only during scientific research, but also in attempts by private individuals and international collaborations.

It is important that governments responsible for pest animal control ensure that welfare concerns are appropriately considered and communicated to the wider community. Australian States and Territories govern animal welfare in comprehensive Legislative Acts. However, in State and Territory legislation, exotic animals may be described as both ‘pests’ and ‘noxious animals’, therefore special consideration of the ethics of eradication programs is required, and the Code should over-ride the legislative requirement to "suppress and destroy noxious animals by any lawful method".

The consequences of not considering the ethical and welfare issues surrounding eradication include risking the support for future eradication programs. Apart from the moral obligation, failure to implement animal welfare practices may inhibit the progress of eradication projects through protest lobbying, denying property access, or even program sabotage (Oppel et al. 2011). Legal proceedings initiated by animal rights activists suspended an eradication attempt for the American grey squirrel (*Sciurus carolinensis*) in the Italian Alps, resulting in the squirrel’s further range expansion (Bertolino and Genovesi 2003). It is of particular strategic importance to widely publicise eradication program actions and objectives before commencement. Nevertheless, engaging with a community, and a variety of stakeholders, can be challenging. In Tasmania, (Section 4.4.2) the fox eradication has been in existence, in one form or another, for over 10 years and yet there is still considerable ongoing debate as to: (1) whether there are foxes; (2) whether money should be spent eradicating foxes; (3) how foxes should be eradicated; and (4) if it is already too late (reviewed in Blackman et al., 2013).

Furthermore, failed eradications may have substantial associated costs with little conservation benefit. A large number of target pests may be killed without achieving the goal of the program. A failed eradication attempt may result in no further management of the pest species, and only a temporary reduction in their impacts. Therefore, it is important that the eradication program is structured in such a way to maximise learning for future operations, wherein a failed eradication may be defended on ethical grounds (Cowan and Warburton 2011).

5.1.7 Documentation

In this report we collated 650 vertebrate pest eradication attempts on Australian offshore islands and mainland areas. Nevertheless, only a small percentage of the AusErad entries contained information for all of the data fields. Most of the information was limited to a simple description of the species, site, and eradication, and it was normal that the record was missing a large proportion of information across the remaining fields. Most commonly, the documented eradication attempts omitted a summary or breakdown of financial cost, detailed methodological descriptions, details of any post-monitoring period, citations to primary or secondary literature and a precise description of the specific location (and habitats) where the eradication was attempted, including geographic coordinates and the size of the area.

Despite the anecdotal importance of stakeholder coordination and socioeconomic factors affecting the success of eradication programs a framework to properly assess and quantify these factors such as “level of coordination”, “degree of public support”, or “quality and experience of the team" was unavailable, and consequently could not be analysed (see also (Pluess et al. 2012a). This issue must be addressed in the future by developing indicators for public support, stakeholder cooperation and the 'experience' of the eradication team. Even socioeconomic factors that are seemingly straightforward to quantify (like eradication effort and annual cost/performance) could not be included in the formal analysis because sufficient detail, and reliable data on eradication costs, resources and manpower, was too often lacking.

Although some missing information can be found or calculated retrospectively, it is preferable that these data are collected by personnel at the time of the eradication attempt. Statistical procedures, which overcome missing data, have been used to impute eradication data that are assumed to be missing, either at random or in a small minority of cases (e.g., Pluess et al. (2012a)). However, using statistical procedures, which allow for missing data without understanding the reasons why they are missing, or failing to examine whether the assumptions made by the statistical procedure are satisfactory, is ill-advised. This is a non-trivial problem, particularly if the results are translated into spurious recommendations for improving eradication success.

5.2 Funding

Costs of eradication attempts were rarely reported for records in AusErad. However, many of the experts contacted during the preparation of this report cited insufficient and inconsistent funding as a major contributor to eradication failure. This is likely to be a common problem because eradication attempts are (perceived to be) expensive. For example, the Lord Howe Island Rat eradication programme is expected to cost in the order of AUD$9 million, and the Macquarie Island rodent and rabbit eradication AUD$24.7 million. Nevertheless, the cost of eradication is often overshadowed by the cost of control, or even doing nothing. For example, it cost over €230,000 to eradicate just 12 Himalayan porcupines *Hystrix brachyuran* from Devon, United Kingdom, but likely prevented much more severe economic losses in the long term, considering the potential impact of the species on crops (Genovesi 2007).

Martins et al. (2006) analysed the cost of eradicating alien mammals from 41 global islands; including three offshore Australian islands. They found that island area alone explained over 70% of the variance in the cost of eradication. Interestingly, the most expensive reported eradication (expressed as US dollars adjusted to year 2003) was the feral cat eradication on Macquarie Island. According to their best (minimum adequate) model, rodent eradications were estimated to be *c.* 1.7 times more expensive per unit area than ungulate eradications (data mostly for goats); see also Figure 1 in Donlan and Wilcox (2007) and their related discussion. For New Zealand eradications, almost half of the data (49%), costs were also lower for more recent eradications, and higher for more remote islands (Martins et al. 2006).

Donlan and Wilcox (2007) observed that variable costs (including local factors) can have a substantial impact on the realised cost of an eradication campaign. The authors listed the most important variable costs of eradication campaigns as including: (1) the needs to mitigate for potential non-target species, (2) the eradication techniques used (e.g. bait stations versus aerial broadcast of bait); (3) the level of local capacity and expertise present, (4) the amount of environmental compliance required; and (5) the levels of bureaucracy. Without considering these costs, eradication planning cannot determine where to best concentrate limited eradication resources.

Rout et al. (2011) provided a case study of black rat invasion on Barrow Island, Western Australia to demonstrate the optimal pre-emptive strategies that defend an island from the impact of a pest that is currently absent. Specifically, they considered the circumstances for which each of the management actions (prevention, detection, control, eradication) are optimal, during an ongoing invasion. Whereas the economic arguments for eradicating a vertebrate pest population can be presented in a formal cost-benefit analysis framework, it is more difficult to present an argument for their eradication to minimise biodiversity loss. Biodiversity serves humanity a range of functions, and important ecosystem services, that are difficult to quantify. Trying to estimate the financial value of these services is complicated, requiring many assumptions and might not even be possible for all ecosystems (Mäler et al. 2008). Aside from ecosystem services, biodiversity can impact mental health with knock-on costs for health spending (Davies et al. 2012) and is thought to decrease healing time and increase productivity by reducing stress levels. Both the number of ways that biodiversity serves human society and the methods used to quantify those services are topics of research in their infancy and undergoing rigorous development.

The other equally important aspect of financing eradication attempts is funding security. Eradications are difficult and subject to many potential problems, including stochastic factors that are outside the control of planners and managers, e.g., the weather. A large contingency margin is required to ensure that any eradication attempt has sufficient funding to see the project through to completion, including the pre- and post-monitoring periods. In particular, the effort (and therefore cost) of an eradication is not stable through time. Because animals at low densities can be more difficult to detect (Russell et al. 2005) cryptic animals can increase the program costs as well as the probability of eradication failure. On Santiago Island in the Galapagos 79,579 goats were removed over 4.5 years for a cost of US$6.1 million (USD$10 - $100 per goat). However, the last 1,000 goats cost $2 million to remove (over USD$10,000 per goat) over 1.5 years (Carrion et al. 2011). Even after eradication has been completed, the monitoring required to confirm the outcome can contribute a substantial additional cost; as much as 10% of the total campaign cost (Cruz et al. 2009).

By its very nature, contingency is difficult to quantify. Ensuring funding security will require policy changes among funding bodies or a government funding stream dedicated to eradication attempts, which would allow rapid access to contingency funding on a particular needs basis. Funding security also requires excellent education/awareness campaigns that are capable of demonstrating the importance of seeing an eradication through to confirmed completion rather than a 'near enough is good enough' approach.

5.3 Prevention is better than cure

Eradication (and control) are measures to remove (or reduce) pest populations in order to minimise their ecological, economic or social impact on the host system. These measures can be expensive and not guaranteed to succeed. An arguably cheaper alternative is to prevent any new species from arriving, or becoming invasive, in the first place. Stringent biosecurity can accrue huge economic benefits, particularly when it is accompanied by species risk assessment screening (Keller and Springborn 2013; Springborn et al. 2011). For example, the Australian Weed Risk Assessment system to screen plant imports provides savings by screening out putative invasive species before they are introduced. Even after accounting for lost revenue from the few non-weeds that might be excluded, screening could save the Australian economy as much as US$1.67 billion over 50 years (Keller et al. 2007). Nevertheless, surveillance can be more effective than absolute quarantine when the impact (and cost of eradication) of a localised population is relatively small (Rout et al. 2011). In this case it is best to catch and eradicate the invasion early rather than attempting to prevent incursion completely.

The number of exotic vertebrate species (not currently established in Australia) detected at large each year has significantly increased over the last decade (Henderson et al. 2011). The majority of new detections were of animals seized, surrendered or stolen from private collections post-border. This suggests that illegal keeping and trading of exotic vertebrates is a significant problem. Alacs and Georges (2008) assessed the illegal wildlife trade in Australia in 1994–2007 and noted that most penalties are substantially lower than the maximum allowable and fines are far less than the black market price of the animals.

Despite Australia's enviable quarantine standards (Keller and Springborn 2013), there is a considerable risk that the next invasive vertebrate pest species is already here; either in private keeping or a native species being traded and transported by human agency beyond its existing range. For example, virtually any species of Australian frog or reptile (with the exception of some threatened species) can be traded or moved to any other part of the country (García-Díaz and Cassey 2014). Currently, Australia applies a risk assessment for non-native species, but an 'open door policy' for species native to somewhere in the country. The result of this approach is that 65% (12 out of 19 species) and 47.1% (32 out of 68 species) of amphibians and reptiles, respectively, introduced in Australia are translocated species (native to a part of Australia and introduced within the country where the species does not occur naturally). There is no a priori reason to think that these domestic exotics would not produce impacts similar to those produced by non-natives upon the recipient communities.

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Appendix A: AusErad Fields

Main fields in the Australian vertebrate pest Eradication database (AusErad)

Field type	Field name	Notes
Taxonomic	Scientific Name	
	Common Name	
	Class	
	Diet	
Introduction	Year of Introduction	
	How population was introduced	
Geographical	Island or Mainland	
	Country	
	State	
	Location (island or park) Name	
	Area_km2	
	Alternate Names	
	Region Name	
	Latitude	
	Longitude	
	Source	
	Elevation_m	
Eradication	Eradication Status	
	Start Year of Eradication	
	End Year of Eradication	
	Reinvaded	
	Connected To other island	
	Primary Eradication Method	
	Secondary Eradication Method	
	Tertiary Eradication Method	
	Min Population Estimation Removed	
	Primary Baiting Method	
	Secondary Baiting Method	
	Eradication Method Details	
	Trapping Method Details	
	Method Description	
	Primary Poison	
	Secondary Poison	
	Cost	
Contact	Contact Name	
	Contact Email	
	Contact Organization	

Appendix B: Australian Eradication Attempts Google Group

In an attempt to reduce any potential reporting bias, we established an internet accessible Google Group called "Australian Eradication Attempts" to facilitate discussion of AusErad records and particular eradication attempts, including any that we could have previously missed. A Google Group is a members-only forum in which members can initiate and contribute to discussions on a particular topic, and can be regulated by forum administrators.

We sent experts invitations to join the Australian Eradication Attempts Google Group and asked them to read and comment on plots and questions posted to the group. All members were given full permissions, allowing them to comment on, create, modify and delete posts. Invitations were valid for a month, after which time they expired and could not be used to join the Google Group, although invitations were re-sent when requested. The list of members invited and joined was visible to all members and invitations were sent to multiple recipients.

Commenting on posts could be done directly from a web browser and required only a computer and internet connection.

We invited all experts for whom we had an email address to become members of the Australian Eradication Attempts Google Group: a total of 38 experts across all Australian States and Territories. In the case of NSW, we invited just one expert because everyone we contacted and invited suggested that particular individual. We do not include contacts in the ACT who were from the Invasive Animals CRC. We also contacted experts for Australia's Commonwealth islands but these individuals were mostly contactable by phone.

The uptake of invitations was disappointing (Figure 26). None of the invited experts from NSW, NT or Qld accepted the invitation and only in SA was the level of participation (among those invited) greater than 50%. All experts had been previously responsive by phone and so the poor level of uptake suggests that email was an ineffective method of communication in this instance.

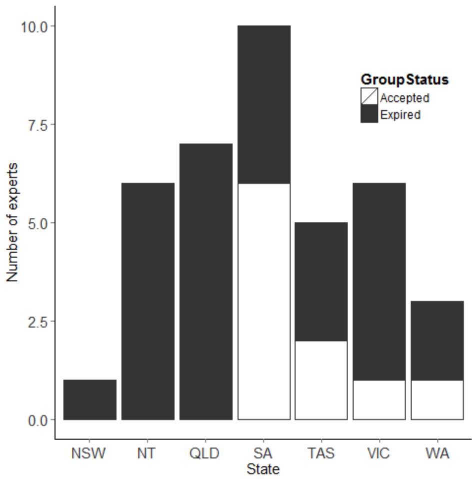

Figure 26: Uptake (or otherwise) of electronic email invitations to become a member of the "Australian Eradication Attempts" Google Group. Experts had a month to respond to the invitation before it expired.

Appendix C: Full global GLM AICc tables

Fixed model terms	AIC*c*	*k*
sqrtTRI_CV	155.4753563	9
sqrtTRI_CV + log10GSArea_km2	156.2835963	10
sqrtElevation_m + sqrtTRI_CV	157.2507408	10
sqrtTRI_CV + PeriodPresent	157.3004945	10
sqrtTRI_CV + log10dSource_km	157.4337725	10
sqrtTRI_CV + log10dgrpMainland_km	157.482344	10
log10GSArea_km2	157.6316363	9
sqrtTRI_CV + log10GSArea_km2 + PeriodPresent	158.1351533	11
sqrtTRI_CV + log10GSArea_km2 + log10dSource_km	158.3870121	11
sqrtTRI_CV + log10dgrpMainland_km + log10GSArea_km2	158.3892584	11
sqrtElevation_m + sqrtTRI_CV + log10GSArea_km2	158.4304806	11
sqrtElevation_m + sqrtTRI_CV + PeriodPresent	158.9749911	11
sqrtTRI_CV + log10dgrpMainland_km + PeriodPresent	159.1996845	11
sqrtTRI_CV + log10dSource_km + PeriodPresent	159.2532978	11
sqrtElevation_m + sqrtTRI_CV + log10dSource_km	159.2551285	11
sqrtElevation_m + sqrtTRI_CV + log10dgrpMainland_km	159.35254	11
log10GSArea_km2 + PeriodPresent	159.3655412	10
sqrtTRI_CV + log10dgrpMainland_km + log10dSource_km	159.504293	11
sqrtElevation_m + log10GSArea_km2	159.6636659	10
log10GSArea_km2 + log10dSource_km	159.7226705	10
log10dgrpMainland_km + log10GSArea_km2	159.783743	10
sqrtTRI_CV + log10dgrpMainland_km + log10GSArea_km2 + PeriodPresent	160.1513001	12
sqrtTRI_CV + log10GSArea_km2 + log10dSource_km + PeriodPresent	160.2398447	12
sqrtElevation_m + sqrtTRI_CV + log10GSArea_km2 + PeriodPresent	160.3321568	12
sqrtElevation_m + sqrtTRI_CV + log10dgrpMainland_km + log10GSArea_km2	160.5312373	12
sqrtTRI_CV + log10dgrpMainland_km + log10GSArea_km2 + log10dSource_km	160.5315566	12
sqrtElevation_m + sqrtTRI_CV + log10GSArea_km2 + log10dSource_km	160.5567716	12
sqrtElevation_m	160.6589241	9

sqrtElevation_m + sqrtTRI_CV + log10dSource_km + PeriodPresent	160.9764883	12
sqrtElevation_m + sqrtTRI_CV + log10dgrpMainland_km + PeriodPresent	160.9838195	12
PeriodPresent	161.169704	9
sqrtTRI_CV + log10dgrpMainland_km + log10dSource_km + PeriodPresent	161.2245934	12
sqrtElevation_m + sqrtTRI_CV + log10dgrpMainland_km + log10dSource_km	161.4077243	12
log10dSource_km	161.4099251	9
log10GSArea_km2 + log10dSource_km + PeriodPresent	161.459202	11
log10dgrpMainland_km + log10GSArea_km2 + PeriodPresent	161.4595556	11
sqrtElevation_m + log10GSArea_km2 + PeriodPresent	161.4928288	11
log10dgrpMainland_km	161.6457059	9
sqrtElevation_m + log10GSArea_km2 + log10dSource_km	161.7664372	11
sqrtElevation_m + log10dgrpMainland_km + log10GSArea_km2	161.8105155	11
log10dgrpMainland_km + log10GSArea_km2 + log10dSource_km	161.8992658	11
sqrtElevation_m + PeriodPresent	162.0849706	10
sqrtTRI_CV + log10dgrpMainland_km + log10GSArea_km2 + log10dSource_km + PeriodPresent	162.3040332	13
sqrtElevation_m + sqrtTRI_CV + log10dgrpMainland_km + log10GSArea_km2 + PeriodPresent	162.3558104	13
sqrtElevation_m + sqrtTRI_CV + log10GSArea_km2 + log10dSource_km + PeriodPresent	162.4576774	13
sqrtElevation_m + log10dSource_km	162.6063281	10
sqrtElevation_m + sqrtTRI_CV + log10dgrpMainland_km + log10GSArea_km2 + log10dSource_km	162.6989987	13
sqrtElevation_m + log10dgrpMainland_km	162.8149929	10
log10dSource_km + PeriodPresent	162.9956355	10
sqrtElevation_m + sqrtTRI_CV + log10dgrpMainland_km + log10dSource_km + PeriodPresent	163.0506235	13
log10dgrpMainland_km + PeriodPresent	163.1278508	10
log10dgrpMainland_km + log10dSource_km	163.5155125	10
sqrtElevation_m + log10dgrpMainland_km + log10GSArea_km2 + PeriodPresent	163.5838926	12
log10dgrpMainland_km + log10GSArea_km2 + log10dSource_km + PeriodPresent	163.5869815	12
sqrtElevation_m + log10GSArea_km2 + log10dSource_km + PeriodPresent	163.6016882	12
sqrtElevation_m + log10dgrpMainland_km + log10GSArea_km2 + log10dSource_km	163.9410253	12

sqrtElevation_m + log10dSource_km + PeriodPresent	164.0368294	11
sqrtElevation_m + log10dgrpMainland_km + PeriodPresent	164.1736305	11
sqrtElevation_m + sqrtTRI_CV + log10dgrpMainland_km + log10GSArea_km2 + log10dSource_km + PeriodPresent	164.530777	14
sqrtElevation_m + log10dgrpMainland_km + log10dSource_km	164.7885363	11
log10dgrpMainland_km + log10dSource_km + PeriodPresent	165.0071732	11
sqrtElevation_m + log10dgrpMainland_km + log10GSArea_km2 + log10dSource_km + PeriodPresent	165.7268409	13
sqrtElevation_m + log10dgrpMainland_km + log10dSource_km + PeriodPresent	166.170091	12

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