

Managing non-target wildlife mortality whilst using rodenticides to eradicate invasive rodents on islands

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Abstract

Invasive rodents are one of the greatest threats to island biodiversity. Eradicating these species from islands has become increasingly practicable in recent decades, primarily using anticoagulant rodenticides. However, this approach also poses risks to native wildlife, and there has been corresponding development in the management of risks to non-target wildlife species. Here we review strategies and tactics used in operational management of non-target risk, using examples from rodent eradication projects conducted on **178** islands where non-target risk assessment and mitigation was a component of the rodent eradication campaign. We identified 17 different tactics within a framework of three strategic approaches: avoidance of risk, minimization of risk, and remediation of the impact of non-target wildlife mortality. We summarize these tactics in terms of their applicability, strengths, and weaknesses for rodent eradication projects in general, plus the potential interactions with achieving rodent eradication. There remains great potential for further innovation in reducing non-target wildlife risks from rodenticide used for invasive rodent eradications on islands, supporting advancement of the social acceptability of the toolset and biodiversity conservation.

Introduction

Rodents account for over 42% of all described mammal species and occur in diverse environments ranging from true desert to mountains, forests and wetlands (Singleton et al. 2006). Four species – *Rattus norvegicus*, *R. rattus*, *R. exulans* and *Mus musculus*—show high adaptability and close association with humans, leading to these taxa becoming invasive in many locations around the world. Rodents commensal with human habitation have well-documented negative impacts including consumption and fouling of human food, damaging infrastructure, and transmitting zoonotic disease (Drake and Hunt 2009). Invasive rodents also impact biodiversity by predation and competition with native fauna and flora (Drake and Hunt 2009), alteration of hydrological regimes, and changed soil chemistry (Simberloff 2009).

Invasive rodent eradication has been achieved for more than 650 islands around the world (Russell and Holmes 2015), eliminating the direct impacts on native fauna and flora and having remarkable conservation benefit (Jones et al. 2016). This approach contrasts with rodent control, which removes a proportion of the population and only achieves positive effects for the time-period the tool is deployed (Cromarty et al. 2002). For a rodent eradication to succeed, adherence to a set of criteria is required, including that mortality exceeds fecundity, every reproductive individual **can access** the removal method, and immigration is **managed to prevent the target species reestablishing on the island** (Bomford and O'Brien 1995; Cromarty et al. 2002). To date, rodent eradication on islands has primarily been achieved through the application of rodenticide, with a handful of very small islands (< 25 ha) achieved by trapping alone (Russell and Holmes 2015). The application of bait has been achieved by broadcast (aerial or by hand) or bait stations, with more than 80% of projects using the second-generation anticoagulant toxicant brodifacoum (Russell and Holmes 2015). Anticoagulant rodenticides block the vitamin K cycle and blood clotting in vertebrate animals, which can lead to hemorrhaging and mortality (Eason et al.

2002). Second generation anticoagulants have been effective for rodent eradications on islands but present risks to other non-target wildlife, through both primary (bait consumption) and secondary (consumption of animals containing rodenticide residues) pathways of exposure (Broome et al. 2017).

Risk to non-target wildlife during rodent eradications on islands is a function of hazard and exposure, whereby hazard is the relative toxicity of the rodenticide to the animal, and exposure is how much and how often an animal is exposed to the rodenticide. Thus, not every wildlife species is subject to the same risk during a rodent eradication operation **nor are all individuals of the same species exposed to the same risk due to either foraging differences between sexes (e.g., giant petrels (*Macronectes* spp.) Gonzalez-Solis and Croxall 2005), or individual feeding preferences (e.g., gulls (*Larus* spp.) Leitch et al. 2014).** Second generation anticoagulants like brodifacoum are known to be acutely toxic to birds and mammals, but less so to reptiles (Herrera-Giraldo et al. 2019; Weir et al. 2015) and there is little evidence of impact to invertebrates (Booth et al. 2003; Craddock 2003). Wildlife behavior will have a key influence on exposure, such as migratory birds only being present on an island for part of the year, or the **marine diet** of nesting seabirds precluding interaction with an exposure pathway on land. *A priori* assessment of non-target risk has been a common feature of many rodent eradication projects. However, to date there has been no attempt to collate information on the strategies or tactics used to reduce non-target risk, nor their relative effectiveness.

There are three basic strategies to mitigate risk to non-target wildlife species during rodent eradications on islands: avoidance, minimization, and remediation. Avoidance involves eliminating the pathway of rodenticide exposure, and thus risk. Minimization entails lessening the risk to the population or individuals by reducing the pathway or hazard. Remediation means compensating for residual negative impacts incurred. Strategies borrow from established mitigation hierarchies (Arlidge et al. 2018). Our goal was to describe tactics used to mitigate non-target risk for wildlife during rodent eradication projects on islands under these three strategies. We achieved this by searching published scientific literature and key practitioner compendiums for actions undertaken during a suite of individual rodent eradication projects. We then summarized applicability, strengths, and weaknesses for rodent eradication projects in general, plus the potential for interactions with achieving rodent eradication (Online Resource 1). **Quantifying the total number of rodent eradication projects that have or have not used non-target mitigation actions is outside the scope of this assessment.**

Methods

We reviewed five resources to identify literature reporting rodenticide-based rodent eradication projects on islands that implemented non-target wildlife mitigation efforts. The first resource, the published proceedings of three symposia addressing eradication of invasive species from islands around the world and a primary reporting outlet for practitioners (Veitch and Clout 2002; Veitch et al. 2011; Veitch et al. 2019). The second and third resources were Web of Science and **Google Scholar, covering publications from 1990–2020**. The keyword search terms for both databases were rodent*, rat, poison*, eradication*, island*, rattus, mus, non-target, nontarget, toxin*, toxicant, brodifacoum (the search was specifically

conducted in the following fashion on Web of Science - TOPIC: (rodent* OR rat OR rattus OR mus) AND TOPIC: (island*) AND TOPIC: (poison*) AND TOPIC: (eradication*) then proceeded to conduct an additional search - TOPIC: (rodent* OR rat OR rattus OR mus) AND TOPIC: (island*) AND TOPIC: (poison* OR toxin* OR toxicant OR brodifacoum) AND TOPIC: (eradication*) AND TOPIC: (non-target OR nontarget)) and for **Google Scholar in the same fashion, but without the word TOPIC**. The fourth resource was a search of all volumes of *Biological Invasions* published between 1999 and 2020 using the same search terms and format. We searched for and highlighted projects only where these projects reported mitigating risk to non-target wildlife species. **For Google scholar, this represented scanning the first 500 titles obtained from each search (2 searches = 1000 titles) and then selecting the function 'add to my library' for those publications that referred to rodenticide-based eradications.** Unpublished accounts known to authors served as a final resource and supplemented our list of projects. Authors provided examples in 2015 and 2020. The literature review of the three published Symposia Proceedings, Web of Science and *Biological Invasions* journal was done in 2015 and again in June 2020 to look for updates, while the **Google scholar search was conducted in 2021 to complement the findings obtained through the other four resources.**

Our goal was not to undertake a comprehensive inventory of all rodent eradication projects but to collate enough examples to describe the main tactics and strategies employed. Within the sourced literature, we identified actions specifically taken to mitigate risk to non-target wildlife, plus operational actions done to maximize efficacy. We also recognized mitigation planning that was advantageous for non-target wildlife, e.g., timing bait deployment to achieve greatest likelihood of eradication success to coincide with predictable absence of migratory birds that may be at potential risk. **Actions such as operational timing, bait type, and bait application method are decisions we expect to be made on every project to optimize efficacy. Likewise, native wildlife species response is an outcome expected on every project.** We also sought to characterize the implications of using these tactics against the goal of achieving 100% mortality of rodents on island. We counted the number of wildlife classes (avian, mammal, reptile, amphibian, invertebrate, fish) that these tactics were used for. We used projects that succeeded or failed at eradicating invasive rodents because this did not affect our goal of identifying tactics.

Results

We identified **145** different sources based on our review. From these sources, we identified **82** different rodent eradication projects (Online Resource 2) covering **178** islands (**i.e., multiple islands located within the same archipelago were treated as the same project**) where measures employed to mitigate non-target risks for wildlife were reported and that we allocated into 17 tactics (Online Resource 1).

The 82 rodent eradication projects ranged in climatic zone (from Tropical to sub-Arctic/sub-Antarctic) and ranged in size (from < 1 to > 100,000 ha). The 82 projects include successful and failed eradications on 178 named islands and numerous unnamed islets **for which non-target wildlife mitigation tactics were documented** (Fig. 1), representing ~ 27% of the 650 islands identified in Russell and Holmes (2015).

These cases (Online Resource 2) addressed risk to non-target wildlife species that are avian, mammal, reptile, amphibian, invertebrate, or freshwater fish (Fig. 2), and to marine environment.

Additionally, we identified 15 sources of information covering 11 future/planned rodent eradication projects on 23 islands or islets (Online Resource 3), for which wildlife non-target mitigation tactics are being considered and documented *a priori*. All tactics proposed in these projects fit within the tactics described and no new tactics were identified.

Below we describe the three strategies: avoidance, minimization, and remediation that cover 17 different tactics for mitigating risk for wildlife species during rodent eradication on islands and provide examples from projects in which those were implemented.

Strategy 1. Exposure Avoidance

Avoidance is concerned with reducing the likelihood of exposure to a rodenticide for the whole or a significant part of the non-target animal's population, either through primary or secondary pathways. Separating non-target animals from sources of exposure needs to consider both spatial and temporal aspects.

Tactic 1. Utilizing predictable, or inducing, absence or inactivity

The timing of rodenticide application is a critical operational consideration to maximize efficacy, particularly in temperate latitudes where bait is applied in autumn or winter when rodent populations are food stressed and bait acceptance will be high, increasing the likelihood of eradication success. This seasonal timing often coincides with natural absence of many non-target wildlife, such as migratory birds, and thus this exposure avoidance tactic is an advantageous artefact of the project. This tactic is the most efficacious for the exposure avoidance strategy because it may not require any additional resources for the operation or interfere with natural wildlife behavior. This tactic is a common feature on most rodent eradication projects. Examples include Palmyra Atoll, **Micronesia** where baiting coincided with adult bristle-thighed curlews (*Numenius tahitiensis*) and four other shorebird species being away at their northern breeding grounds (Wegmann et al. 2012), and Campbell Island, **New Zealand** when brown skuas (*Stercorarius antarcticus*) were absent (McClelland 2011). This tactic can exploit seasonal torpor or aestivation for species that do not leave the island. For example, on Pinzon Island, **Ecuador**, endemic land snails (*Naesiotus* spp.) were aestivating at the time rodenticide was applied (Rueda et al. 2019). This may also explain the lack of documented impacts to New Zealand reptiles—which are mostly inactive during cooler months (Towns and Broome 2003, Griffiths and Towns 2008; Wilkinson and Priddel 2011)—from the numerous rodent eradications, generally completed over the winter.

While this tactic has been successful for many species, it is not immune to uncertainty. On Hawadax (previously Rat) Island, **United States**, rodenticide was applied during boreal autumn with subsequent winter months being a period when bald eagles (*Haliaeetus leucocephalus*) were historically absent or in low numbers (Buckelew et al. 2011). However, higher than anticipated numbers of eagles were on the

island post-bait application, possibly in response to carrion availability or weather conditions, and succumbed via this secondary exposure pathway (Buckelew et al. 2011).

Where seasonal absence or inactivity cannot be utilized, inducing temporary dispersal of animals may be an option (Gorenzel and Salmon 2008). On Allen Cay in the Bahamas, nest destruction and egg removal occurred prior to bait application being implemented to **push** breeding birds to nearby islands (Alifano et al. 2012). On South Georgia Island, **South Georgia and the South Sandwich Islands**, attempts to encourage an early migration of breeding adult brown skua (*Stercorarius antarcticus*) **were implemented** by oiling eggs to prevent them from hatching (Martin and Richardson 2019). However, this approach will be unsuitable for island endemics or species with reduced dispersal ability, and it carries the risk that animals repelled may return or move to other sites within the same island.

Tactic 2. Captive holding in situ

Non-targets can be isolated from possible access to toxicant by holding them temporarily and securely on their home island(s). This may be appropriate for non-target species deemed at risk and present year-round. Examples of this tactic on previous projects include Northern tuatara (*Sphenodon punctatus punctatus*) on Little Barrier Island, **New Zealand** (Fisher et al. 2011), two species of reptiles on Desecheo Island, **Puerto Rico** (Will et al. 2019), and deer mice (*Peromyscus maniculatus anacapae*) on Anacapa Island, **United States**, where individuals of each species were captured, then housed in captivity during and after bait application (Howald et al. 2009).

There are risks inherent in bringing wild-caught animals into captivity (Animal Care and Use Committee, American Society of Mammalogists 1998) and being able to adequately maintain them for a required duration. Thus, a critical requirement of this tactic is that captive husbandry of the species is well enough understood to ensure good health until release, particularly recognizing the stress caused to animals by capture and confinement. If this information is not available, its collection should be an important part of the feasibility assessment, as done **in Australia** with Lord Howe Island woodhens (*Gallirallus sylvestris*) and currawongs (*Strepera graculina crissalis*) (Wilkinson and Priddel 2011); with friendly ground dove (*Gallicolumba stairifor*) for the implementation of Nu'utele and Nu'ulua in the Samoa (Butler 2005); and five species of Galapagos finches (*Geospiza fuliginosa*, *G. fortis*, *G. scandens*, *Camarhynchus parvulus*, *C. pauper*) and Galapagos short-eared owls (*Asio flammeus galapagoensis*) for the planned Floreana Island, **Ecuador**, rodent and feral cat eradication (PAC unpublished data).

It is necessary to decide how many individuals to put into captivity **early in planning processes. Numbers of individuals** will be influenced by logistics including how many can be functionally accommodated given resource availability. Released individuals may play a critical role in securing population persistence after the eradication (e.g., Anacapa deer mice; Howald et al. 2009), and thus, understanding **minimum viable** population sizes and potential loss of genetic variability is also essential (e.g., Pergams et al. 2000). A further decision **required during planning** is when to release animals back into the wild population. Historically, this has included estimations of when the environment is safe, or when bait is no longer visible. However, this time should ideally be when the pathway of exposure is no longer present or

biologically significant. This can be achieved by measuring biologically relevant indicators, such as conspecifics in the wild (Brooke et al. 2012), phased releases using conspecifics as sentinel species (e.g. Galapagos hawks (*Buteo galapagoensis*) on Pinzon Island, **Ecuador**, PAC unpublished data), or other biological indicators such as prey items (e.g. lava lizards *Microlophus sp.*) as on Pinzon Island (Rueda et al. 2016; Rueda et al. 2019).

We note that captive holding *in situ* may not necessarily achieve complete isolation of a population if pathways can interact with captive held animals, e.g. as has been recorded in zoos where invertebrates and house mice (*Mus musculus*) consuming rodenticide baits entered cages leading to secondary poisoning of captive animals (Hernandez-Moreno et al. 2013).

Tactic 3. Captive holding ex situ

Holding a population of at-risk individuals in captivity off-island may be preferable if on-island holding is not practical or appropriate. For example, on Plaza Sur Island, **Ecuador**, 40 of an estimated 400 land iguanas (*Conolophus subcristatus*) were captured and taken to nearby Santa Cruz Island, **Ecuador** where suitable facilities and technical personnel were available (Campbell et al. 2012). Similarly, 15 giant tortoises from Pinzon Island, **Ecuador** (*Chelonoidis ephippium*) were taken to Santa Cruz Island, **Ecuador** and held for up to two years after rodent eradication baiting (Jensen et al. 2015). **As part of the Kapiti, New Zealand, rodent eradication, safeguard populations of 243 weka (*Gallirallus australis*) and 66 North Island robins (*Petroica australis longipes*) were captured and successfully held ex situ for the duration of the campaign** (Towns and Broome 2003). An important consideration with this tactic is reducing the likelihood of spreading new invasive species **and diseases** by moving animals to a host site, and again back to their home island. For Plaza Sur, **Ecuador**, this included protocols for minimizing internal and external parasite and seed transfer between islands (Campbell et al. 2012). Determining how many individuals, and when to return captive individuals to the home island, is a key consideration as described for *in situ* holding.

Tactic 4. Conservation translocation off-island

For some projects, it may be appropriate to establish a permanent **additional** population at a separate site using individuals from the project island. As with *ex situ* holding, this requires capture and transport of individuals to a new site. However, unlike *ex situ*, the intent is to create a self-sustaining population in the wild and not in captivity. **Conservation translocation is the intentional movement and release of a living organism inside or outside its indigenous range with a conservation benefit objective** (sensu IUCN 2013). **Conservation translocation** is applicable where the cause(s) of population decline are either unknown or unable to be addressed in a suitable time frame, or where animals are unable to re-establish a population through dispersal after threatening processes have been addressed. It can safeguard a population, establish additional populations and act as a source for reintroductions (Cunningham 1996; Dickens et al. 2009; Woodford and Rossiter 1994). **Nonetheless, it is important to acknowledge that the remaining portion of the source population might be rendered unsustainable in the long term due to non-target mortality, and/or because of being overharvested for translocation and creation of an insurance**

population off-island. Additionally, it is axiomatic that any receiving location for a translocated population must provide enough suitable habitat and be free of **those** threats (e.g., predators, competitors, disease) **that caused the original decline.** In practice, however, these conditions may not be easily satisfied. On Codfish Island, **New Zealand**, fernbirds (*Bowdleria punctata wilsoni*) were translocated to create an 'insurance' sub-population on rat-free Kaimohu Island, **New Zealand**; though this translocated population ultimately failed to establish, possibly because of the island's size (12 ha) and lack of suitable vegetation (McClelland 2002). A further 21 Codfish Island fernbirds were translocated to Putauhinu Island, **New Zealand** (148 ha) after kiore (*R. exulans*) were removed. The Putauhinu fernbird population became self-sustaining and would have provided a source population had reintroduction to Codfish Island been necessary (McClelland 2002). Other examples are **Breaksea Island, New Zealand, where South Island robins (*Petroica australis*) were transferred to Hawea, New Zealand establishing the densest population of the species in the region and dispersing across open water to nearby Wairaki Island (Thomas 2002), and Selvagem Grande, Portugal, where a small population of Berthelot's pipit (*Anthus bertheloti bertheloti*) was translocated to Selvagem Pequena, located 22km away, to establish a safeguarded population in a place where the species had been reported previously (Olivera et al. 2010).**

In most instances, this conservation translocation off-island will be required to be combined with captive holding *ex situ* or *in situ* depending on availability of appropriate facilities to reduce the likelihood of spreading new invasive species by moving animals to host site or until enough numbers are captured. An example of the former was the translocation in 2019 of more than two thousand land iguanas (*Conolophus subcristatus*) from Seymour Norte Island, **Ecuador**, to Santiago Island, **Ecuador**, as part of the Seymour Norte Island rodent eradication, during which land iguanas spent at least a month on Santa Cruz Island, **Ecuador** (where housing facilities were available), prior to being released on Santiago Island (PAC unpublished data). For Vahanga Island, **French Polynesia** project, Polynesian ground-dove (Tutururu; *Alopecoenas erythropterus*) and Tuamotu sandpiper (Titi; *Prosobonia parvirostris*) were placed in captive holding *in situ* for a couple of hours to days prior to being translocated to Tenararo Island, **French Polynesia** (Pierce et al. 2015). A consideration when translocating birds off-island is the possibility of birds returning to their home island. This was observed for Titi during the Vahanga project, where some individuals translocated to Tenararo **flew** back to Vahanga, even though their outermost primaries were plucked on both wings (1–3 feathers per wing) prior **to** release (Pierce et al. 2015).

Strategy 2. Risk Minimization

Minimizing the potential for primary or secondary exposure is a strategy frequently employed to reduce mortality in a non-target population. **This can be implemented if complete avoidance of non-target exposure to a rodenticide is not feasible and can also be combined with remediation tactics.**

Tactic 5. Choice of rodenticide

Toxicant profiles differ between rodenticides and have significant impact on operational efficacy and present different toxicity hazards to non-target wildlife. Three primary classes of rodenticide have been used in island eradication attempts (DIISE 2014; DIISE 2018; Parkes et al. 2011). Second generation

anticoagulants (coumarins - particularly brodifacoum) are widely used, followed by 1st generation anticoagulants (coumarins, indandiones), with a handful of projects using acute toxins (zinc phosphide, strychnine) (Howald et al. 2007). Second generation anticoagulants are more potent **than first generation anticoagulants**, requiring only a single feed to be lethal. They are thus more efficacious in achieving rodent eradication but present a higher non-target risk. First generation anticoagulants are less toxic **than second generation anticoagulants** and require multiple feedings to achieve a lethal effect in rodents, but present less risk to non-targets accordingly (Howald et al. 2007). Rodenticide choice can have clear implications for operational efficacy, with projects that use the most common 2nd generation anticoagulant (brodifacoum) having a lower failure rate than projects using the most common 1st generation toxicant (diphacinone) (Parkes et al. 2011). Examples of first-generation anticoagulants being used to reduce risk to non-targets include Nishi Island, **Japan** (Hashimoto 2010), and **Cocos Island, Guam, where it was used to reduce risk to native forest birds** (Lujan et al. 2010). **Regulations at the time of implementation stipulated that diphacinone was the only rodenticide authorized for rodent control and eradication operations on Lehua Island, United States (Parkes and Fisher 2017), but (Pitt et al. 2011) the project failed to remove rodents.**

Tactic 6. Reduce attractiveness of bait to non-target species

Bait used in rodent eradications is typically cereal-based compressed pellets or blocks with well-established palatability to targeted rodents. Such bait can be palatable to other species, creating potential for primary exposure. Green or blue colored bait is thought to make it less attractive to some birds (e.g. Buckelew et al. 2011; but see Hartley et al. 1999). Color may need to be tailored to specific species of conservation value (Hartley et al. 1999; Weser and Ross 2013). Using bait blocks (28g average weight) instead of bait pellets (1-2g average weight) may protect small animals from toxicant ingestion, but blocks may take longer to degrade and when they do so, may be ingested. Further, blocks may increase exposure durations for invertebrates that consume them, increasing duration of exposure to their predators through secondary pathways.

Bitrex in rodent baits was proposed as a repellent for wildlife (Meier and Varham 2004). As bitrex also repels rodents, subsequent practice considered that this presented an unacceptably high risk of operational failure for rodent eradication projects (Cromarty et al. 2002). We found one account proposing the use of bait repellent for deer (Russell KJ et al. 2017) during the future/planned Stewart Island, New Zealand rodent eradication.

Tactic 7. Bait application method

Three main methods of bait application exist: bait station, hand broadcast and aerial broadcast, and represent a critical consideration to achieve operational efficacy (Howald et al. 2007). Aerial broadcast is typically the only effective method available for larger islands (~ > 200 ha), with a few exceptions like Langara Island (3,137 ha) in Haida Gwaii, **Canada**, from which rodents were successfully removed by using bait stations (Taylor et al. 2000). Broadcast application—by air or by hand—requires rodenticide distribution across the environment, thereby creating potential primary exposure pathways for non-target wildlife. In contrast, on islands where feasible, bait stations can ensure that baits are accessible only, or

principally, to the targeted wildlife (Pitt et al. 2011). For example, on Monito Island, **Puerto Rico**, native birds were excluded using a simple PVC tube (García et al. 2002) with an aperture that would admit rats but not larger birds. On Bird Island, **Seychelles**, access to bait within bait stations by skinks and land crabs was reduced by elevating the stations and modifying the entrance (Merton et al. 2002). New bait station designs have exploited differences in morphology and behavior of target and non-target wildlife species, but their use may not always be feasible (e.g., very young rats that cannot climb into a bait station) or where target species and non-target wildlife have very similar size range and behaviors. Bait stations must also be robust to the wildlife or domestic species present on an island, e.g., bait stations on Gunner's Quoin, Mauritius, were damaged by cattle (Bell 2002).

Bait stations have been used in combination with aerial broadcast to minimize localized risk to non-targets. During the aerial broadcast for Anacapa Island, United States, a 15ha exclusion zone was applied around rufous-crowned sparrow (*Aimophila ruficeps obscura*) habitat and bait stations used within this exclusion zone. Despite this measure, monitoring indicated significant mortality in rufous-crowned sparrows (Howald et al. 2009). Similarly, localized use of bait stations was unsuccessful at preventing significant mortality in fernbirds on Codfish Island, **New Zealand** (McClelland 2002). This failure was attributed to birds within the bait station zone moving into areas where bait was broadcast following the mortality of birds holding adjacent territories (McClelland 2002). To reduce the likelihood of globally threatened wandering albatross (*Diomedea exulans*) chicks interacting with bait, pellets within a five-meter radius of **all nests on Macquarie Island, Australia**, were removed **after aerial baiting passes** and replaced with bait stations (Springer and Carmichael 2012).

Aerial broadcast may also present an exposure pathway to aquatic animals, both in marine and freshwater ecosystems. This exposure can be minimized by designating exclusion zones or using hand broadcast or bait stations in these areas (Lord Howe Island Board 2019) or by suspending rodent baits in drainage systems to reduce likelihood of bait entering water as done on Wake Island, **United States** (CCH unpublished data). Physical methods used include plastic sheeting over inland water bodies such as those implemented during the Bird Island, **Seychelles** application (Merton et al. 2002). On Anacapa Island, **United States**, a deflector shield was fitted to the bait bucket to guide rodenticide broadcast to one side of the aircraft and improve accuracy near water bodies (Howald et al. 2009), a method now commonly used.

Tactic 8. Optimizing bait application rates, bait availability, and use of other techniques to reduce rodenticide use

Pott et al. (2015) proposed a standardized process for determining optimal baiting rates specific to conditions present on any given island using only the amount of bait required to achieve a successful rodent eradication (i.e., available for a minimum of four nights during each application), while minimizing the risk of non-target exposure and mortality. For some projects, it may be possible to combine rodenticide application with techniques to reduce the amount of toxicant applied or available to non-targets, and therefore, reduce risk to non-target wildlife. **Pascal et al. (1996) provide an example where a rodent eradication on eight islands located in the Brittany Channel and Cancale archipelago, France, and**

four islets off the coast of Martinique (Pascal et al. 2004), used a two-stage approach with trapping followed by a hand-baiting operation to reduce total amount of toxic bait input into the environment. On Quail Island, New Zealand, a similar approach was implemented; but there, trapping was followed by an intensive ground poison operation using bait stations to reduce non-target wildlife poisoning (Kavermann et al. 2003). This tactic has the potential to significantly interact with operational efficacy because the goal of 100% rodent mortality cannot be compromised to achieve eradication success. **Bait formulation can also play an important role in mitigating bait availability to non-target wildlife. Bait can be developed to withstand environmental conditions, thereby securing exposure to all rodents while also breaking down faster with rainfall and reducing non-target wildlife exposure. For Codfish Island, New Zealand, this was a consideration a priori (Broome 2009) and is now a commonly used method.**

Tactic 9. Stagger bait deployment

Staggering bait deployment implies deploying rodenticide in spatially distinct units at different times so non-target species have differential exposure to bait. This tactic was used to ensure the presence of viable endemic deer mouse populations throughout the eradication of rats on Anacapa Island, **United States** (Howald et al. 2009). Anacapa is composed of three closely grouped islets and managers staggered the rodenticide application over two years, treating only one island first, then the remaining two 12 months later. Risk of reinvasion of the treated islet during this 12 month period was avoided by using covered bait stations between islets to intercept any dispersing rats (Howald et al. 2009). Eradication on South Georgia, **South Georgia and the South Sandwich Islands**, was a phased operation undertaken over multiple years, in part to reduce non-target risk, **as an artefact of the island being divided by glaciers into rat-present and rat-absent sectors**. Thus, only segments of the South Georgia pintail (*Anas georgica georgica*) **and South Georgia pipit (Anthus antarcticus)** populations were exposed to risk in any one year (Martin and Richardson 2019), allowing assessment of non-target wildlife impacts before moving on with the next project phase.

Tactic 10. Strengthen resilience of non-target population

Many external influences may threaten, or at least reduce, the viability of native vertebrate populations. If those threats can be eliminated or reduced, it may minimize the threat posed by a toxicant baiting campaign to the extent that the campaign becomes possible. A non-target population may be made more resilient to a toxicant baiting program by increasing its numbers through a management action, such as providing protection from predators, head-starting, supplementary feeding, population supplementation from genetically compatible populations that exist elsewhere, treatment of parasites with antiparasitic or restoration of habitat. Having a suitably resilient non-target population was a pre-requisite for at least one project. The Macauley Island, **New Zealand**, rat eradication could not proceed until it was confirmed that the Kermadec parakeet (*Cyanoramphus novaezelandiae cyanurus*) population inhabiting the island was **a minimum of 3000 individuals**, because this population was the most significant in the Kermadec Island group, **New Zealand** (Greene et al. 2014). This example shows a passive approach to this tactic. For Floreana Island, **Ecuador**, however, this has been actively considered. Darwin's finch—especially Floreana medium tree-finch (*Geospiza pauper*)—nests will be injected with a permethrin insecticide against the

parasitic fly *Philornis downsi* prior to and immediately after rodent eradication to strengthen the resilience of these populations by increasing nestling survival (Digby et al. 2020).

Tactic 11. Provision of alternative food sources

Increasing the availability of alternative, highly preferred food to non-target animals may mean they are less likely to find and consume toxic baits. On Plaza Sur Island, **Ecuador**, fallen *Opuntia* sp. cladodes are readily eaten by land iguanas (*Conolophus subcristatus*) and appear to be a major food source (Campbell et al. 2012). During an operation to eradicate mice there, iguanas were supplied with *Opuntia* cladodes collected on a nearby island. Since *Opuntia* reproduces vegetatively as well as sexually, care was taken to remove uneaten cladodes to preserve the genetic integrity of the Plaza Sur *Opuntia* population (Campbell et al. 2012). While there was no monitoring of supplementary feeding effectiveness after toxic bait was applied, of an estimated 400 land iguanas, nine were discovered dead, seven with bait in their stomachs (W. Tapia pers. comm.). This tactic has two important potential drawbacks. Supplementary food may provide an alternative food source to targeted rodents, reducing the chance they will consume toxic bait. For this reason, uneaten *Opuntia* cladodes were removed at the end of each day on Plaza Sur, **Ecuador**. It may also attract more non-targets to the eradication area. For example, during operations on Bischoff Island, **Canada**, deer carcasses were put on neighboring islands to lure bald eagles away, however, the risk that this action may attract more eagles to the larger region was acknowledged (PJM unpublished data.). On Shiant Isles (Garbh Eilean and Eilean an Taighe), **Scotland**, rabbit carcasses were provided on tables for eagles as a diversionary food source to reduce scavenging of rodents and non-target carcasses. However, no consumption of rabbit carcasses or eagle mortality was observed during the time carcasses were available (Main et al. 2019).

Tactic 12. Reduce quantity or availability of toxicant by secondary exposure pathways i.e., pathways other than bait

Secondary poisoning is a pathway whereby scavengers or predators feed on carcasses or sub-lethally poisoned animals. Reducing or eliminating that exposure pathway can be achieved by collecting carcasses of poisoned target animals. On Macquarie Island, **Australia**, during **both the first and second** rodent and rabbit eradication attempts, collection teams patrolled the island after baiting, burying rabbit carcasses to reduce non-target risk to scavenging birds (Springer 2016). While most carcasses found were already scavenged, suggesting **efficacy of this tactic is limited** (Springer 2016; Springer and Carmichael 2012), the overall non-target mortality (and a function of all tactics utilized) during the second successful attempt (when 100% of the island was treated) was 1,464 birds, compared to 960 carcasses when only 8% of the island was treated in the first incomplete attempt. **The primary difference between the two attempts lies in the elimination of the majority of rabbits with rabbit hemorrhagic disease prior to baiting in the second removal effort** (Springer 2016).

On Rangitoto and Motutapu Islands, **New Zealand**, attempts were made **to protect shorebirds** by reducing the density of amphipods via the removal of beach organic matter on which they feed (*Talorchestia* spp.; e.g. Dowding et al. 2006). Amphipods were identified as a potential pathway via which New Zealand

dotterels (*Charadrius obscurus*) could be exposed to rodenticide (Griffiths and Towns 2008). **As a result of this project, it was identified that by implementing this action for future rodent eradications on New Zealand, coupled with removal of baits from the high-water mark area immediately after each bait drop, mortality for dotterels is expected to reduce considerably** (Dowding et al. 2006).

On Murchison and Faraday Islands, **Canada**, prior to rodent eradication, removal of Sitka black-tailed deer (*Odocoileus hemionus sitkensis*) through planned culls was conducted. This was done to minimize risk of secondary poisoning for native wildlife species such as bald eagles, common ravens (*Corvus corax*), black bears (*Ursus americanus*) and other wildlife species that could scavenge deer carcasses (Gill et al. 2014). **Once the eradication was completed, after 418 hours of carcass search time, common ravens (n = 11) were the only scavenger species recovered on Murchison and Faraday.**

Tactic 13. Provision of treatment for poisoning

Administration of Vitamin K₁ is an effective treatment for individuals with anticoagulant rodenticide poisoning (Watt et al. 2005) and to prevent poisoning when given prophylactically. Where individual non-target animals with poisoning can be identified and captured, treatment for anticoagulant poisoning has frequently been successful. On Palmyra Atoll, **Micronesia**, captured bristle-thighed curlews were successfully treated by injections of vitamin K₁ after ingesting brodifacoum (GRH unpublished data). On Henderson Island, **Pitcairn Islands**, Henderson rails (*Porzana atra*) exposed to brodifacoum were successfully treated the same way in two of eight cases (Brooke et al. 2012). Oral vitamin K₁ was used successfully with surviving Galapagos hawks that were captured after exposure to toxicant through prey consumption, showing clinical signs up to six months after a rodent eradication project was completed on Pinzon Island, **Ecuador** (Rueda et al. 2019).

For eradication operations, prophylactic distribution of Vitamin K to non-target populations is limited due to **the absence of** a practical mechanism to deliver appropriate doses reliably and repeatedly to non-target wildlife while not exposing target rodent species. On Pinzon Island, **Ecuador**, it was unsuccessful when administered injected into supplementary food (goat meat) to protect Galapagos hawks from poisoning after being released back onto Pinzon Island, **Ecuador** (Rueda et al. 2016; Rueda et al. 2019). This treatment was also used on smaller Selvagem Grande, **Portugal**, with Vitamin K₂ added to drinking water in elevated stations where Berthelot's pipit could access it but house mice and rabbits could not (Olivera et al. 2010). Birds were observed using the platforms, but the efficacy could not be determined. Pipits showed a decline during the operation year but rebounded shortly after (Olivera et al. 2010).

Strategy 3. Remediation

A third strategy for managers is remediation of impacts. Expected losses from non-target impacts may be offset from the longer-term gains expected from natural immigration and recovery in a rodent free environment. If extirpation **or substantial reduction** of a non-target population is a possibility, an additional tactic is reintroduction of the species back to the island.

Tactic 14. Natural immigration

Where the impacted non-target species is mobile and prone to disperse, sufficient population replenishment or reestablishment may be achieved by natural immigration of off-island wild conspecifics into vacant territories. This tactic was deliberately applied *a priori* on Pinzon Island, **Ecuador**. There, managers judged that Galapagos endemic short-eared owls (*Asio flammeus galapagoensis*), were present on sufficient surrounding islands that they could easily recolonize Pinzon, if their population was extirpated by poisoning (Fisher and Campbell 2012). Observations three years post-rat eradication confirmed that owls have returned to Pinzon (FC unpublished data), though there was an owl death from poisoning two years after baiting (Rueda et al. 2016). When immigration occurs, the duration of risk from residual toxicant and the timing of immigration will determine the fate of immigrating individuals. On Pinzon Island, **Ecuador**, juvenile and sub-adult hawks that had been off-island returned within days of territory-holding adults being brought into captivity. Prior to documenting this immigration, the Pinzon population was believed to be a closed population. Most of the immigrating individuals were captured and held for several weeks, avoiding risk at that time.

When assessing whether to rely on this strategy, managers will need to take into account ecological criteria such as the population genetics, size of the animal, its dispersal ability, the availability of biological corridors within the landscape matrix over which the animal must disperse to reach a suitable and vacant territory, the distances of that dispersal, and the quality of the habitat into which immigrants move (see e.g. Hilty et al. 2006). This tactic also carries the assumption of successfully eradicating rodents, and managers should evaluate the possible consequences for non-target populations should failure occur.

Tactic 15. Natural recovery

Where the non-target species population declines but is not extirpated and where invasive rodents are absent, populations typically increase in abundance, often spectacularly (Jones et al. 2016). For native predators, their prey bases need to recover before their population can increase; as such, predator populations often see a time-lag in recovery (e.g., Croll et al. 2016). This tactic is often applied in conjunction with other tactics and is widely applied by managers, although infrequently described as an explicit tactic during planning.

On Pinzon Island, **Ecuador** this tactic was applied *a priori* for six species of Darwin's finches, where a natural recovery was expected from an increase in breeding success upon rodent removal. There, managers based their decision on monitoring data collected from previous successful rodent eradication projects conducted in the archipelago and where the same or similar species were present (Fisher and Campbell 2012). **On Barrow and Middle Islands, Western Australia, this tactic was applied for Barrow Island mouse (*Pseudomys nanus ferulinus* and *Zygomys argurus*) populations, which were not able to be excluded from bait stations; as a result, they suffered a decline during and immediately after rodent eradication. Native mouse population levels returned to pre-baiting levels within 12 months (Morris 2002).**

Tactic 16. Reintroduction

IUCN guidelines define reintroduction as the intentional movement and release of an organism inside its indigenous range, from which it has disappeared (IUCN 2013). This clearly includes re-establishing a population that may have been extirpated during a rodent eradication. **As such this tactic can be combined with Tactic 4, whereby conservation translocation off island creates a potential source of appropriate founders for future reintroduction.** Reintroduction science has progressed rapidly in the last 10 years and there are now guidelines (e.g. IUCN 2013) and the IUCN has a specialist working group devoted to reintroductions (<http://www.iucnsscrg.org/>). There are thus precedents to follow and lessons learned from previous attempts (e.g. Armstrong et al. 2015; Soorae 2013). We are aware of two wildlife reintroductions completed after extirpation during a rodent eradication specifically because of non-target impacts. Deer mice were taken from Middle and West Anacapa Islands, **United States**, to East Anacapa (Howald et al. 2009). During the Boodie Island, Australia, eradication project, burrowing bettongs (*Bettongia lesueur*) were extirpated and later reintroduced. Individuals used for this translocation came from Barrow Island, **Australia** (Morris 2002). **Reintroduction was successful with an estimated population of 200–300 individuals in 2002 resulting from 36 animals translocated in 1993.** On Codfish Island, **New Zealand**, reintroduction was considered a *priori* in case fernbirds were eradicated completely from the island during the rodent eradication; however, it was not required as **sufficient** fernbirds capable of reestablishing the population survived (McClelland 2002).

When considering implementing this tactic **it is** important to consider genetic provenance of founder populations and **to fill associated** knowledge gaps (e.g., potential genetic uniqueness) prior to implementing rodent eradications. Additionally, managers should also estimate the size of any source population and the biological significance of toxicant persisting in the landscape.

Tactic 17. Reinforcement

Small, isolated populations are inherently more vulnerable to extinction from internal population malfunction and external threat than large ones, even if the small population is composed of healthy, out-bred individuals (e.g. Fryxell et al. 2014). Reinforcement aims to safeguard a wild population by preventing it becoming so small that it suffers this increased risk of extirpation. **But it can also be used to adjust or enhance demographic structure or genetic representativeness of the recipient population.** Reinforcement is where organisms are intentionally moved and released into an existing population of conspecifics (IUCN 2013). Many of the considerations applicable to Tactic 16 also apply here. In Tactic 16, however, the population has been extirpated, while in the case of Tactic 17 (reinforcement), individuals remain *in situ*. There are also similarities to Tactic 10, which involves increasing resilience of the population *before* an event. This is particularly applicable when the risks of keeping a captive population for an indefinite period are too high, or where moving the animals off-island is unacceptable for biological, cultural, or political reasons. **Additionally, this tactic can be combined with Tactic 4, whereby conservation translocation off island creates a potential source of individuals for future reinforcement of the remaining in situ population or with Tactic 14, where individuals that arrive via natural dispersal and occupy vacant territories can be supplemented with a reinforcement tactic.**

The best example from our review for reinforcement is from Pinzon Island, **Ecuador**, where fifteen adult endemic giant tortoises were brought into captivity for a total of four years (two years prior to bait application until two after bait application was concluded) (Rueda et al. 2019) to be part of the fifty-year head-starting program led by the Galapagos National Park. **For this program**, eggs or recently hatched individuals were brought into captivity to be reared until 4–5 years old, at which time they were released back to Pinzon Island, **Ecuador**, to reinforce the wild population and prevent the species' extinction (Jensen et al. 2015; Rueda et al. 2019).

Discussion

Implicit in our review is the principle that eradication operations will only be progressed after the cost-benefit of attempting eradication, including evaluation of non-target risk (Broome et al. 2014; Keitt et al. 2015), indicates that the cumulative benefits to native fauna outweigh the cumulative costs. *A priori* consideration of non-target risk is necessarily undertaken on an island-by-island basis with reference to the fauna present and the potential pathways that might expose non-targets to rodenticide. Such assessments frequently identify information gaps which confer high uncertainty to risk estimates. These can often be addressed by field research into the likely exposure of non-target wildlife to non-toxic bait. While the main consideration determining the amount of rodenticide applied is assurance that all targeted rodents will be killed, there is a balance in also minimizing risk to non-targets. Also, on an island-by-island basis, the acceptability of non-target risks is gauged by biological and socio-political factors, where unacceptable risk must be managed.

Co-design of non-target mitigations and eradication operations need to consider the implications of decisions on the likelihood of eradication success and non-target impacts. As they are interconnected, managers need to balance this tension in project design, while ensuring that eradication success is maximized. Eradication failure results in no benefits for threatened species, non-target losses that may not be recouped, and the continuation of impacts of invasive rodents.

Assessing risk to non-target wildlife is a vital part of rodent eradication planning, implementation, and follow-up on islands. In this review, we identified 82 projects using one or more of the 17 tactics to manage potentially harmful outcomes to non-target wildlife during 82 rodent eradication projects implemented on 178 islands and 11 future/planned rodent eradication projects on 23 islands considering *a priori* implementation of one or more of these 17 tactics to manage non-target wildlife risk. These results were sufficient to achieve our goal of defining and describing non-target mitigation but should not be considered a comprehensive accounting of all projects deploying these strategies and tactics. **There is a need for greater reporting in conservation** (Sutherland et al. 2004, Pullin et al. 2020), **particularly of failures** (Catalano et al. 2019). **While invasive mammal eradications on islands represent one of the more reported endeavours** (Baker and Bode 2021), **non-target wildlife outcomes require greater reporting in publicly available literatures to ensure lessons learned are transferred.**

Producing a reliable risk assessment requires consideration of a range of factors. These include, but are not limited to, all possible pathways by which non-target animals could be exposed to toxicants; the duration and magnitude of any such exposure; whether the population is an evolutionary significant unit (any case where the risk to an island endemic species or evolutionary significant unit is present requires extra caution); the species involved and its phylogeny, or if that is not well known, its genetic profile; the size of the non-target population; its susceptibility to poisoning; and its conservation status. Significant information gaps should be identified and used to guide additional input, including expert assessment, field studies, or trials. In some cases, implementing management actions to alleviate the risk (e.g., captive holding) may be more **cost effective** than attempting to fill the knowledge gap. Managers should also be mindful of the dynamic nature of biological systems and employ monitoring to detect changes in a timeframe that allows adaptive management.

These strategies and tactics provide a suite of options for managers of rodent eradication projects to consider. Tactics should be matched to the risks identified and we expect that, like previous projects, most projects will only employ a subset of these tactics. No-action decisions for identified risks are also an option, and like the tactics, will have consequences that should be factored into the decision-making process.

Continued research and development to mitigate risks to non-target species and populations is expected to increase the utility of rodent eradication as a conservation strategy. This includes increasing the efficacy of existing tactics, such as bait deterrents for reducing bird and invertebrate interaction with bait (e.g. Clapperton et al. 2015), and developing new tools, such as rodent specific toxicants (Hartley et al. 1999; Weser and Ross 2013) and genetic biocontrol (e.g. Campbell et al. 2015), which could eliminate non-target risks for many species altogether by only presenting a hazard to invasive rodent populations. Waiting for such developments to be realized may not be prudent given the severity of risk that invasive rodents pose on many islands. Our review demonstrates the range of practical measures available for reducing risk to non-target wildlife during rodent eradications, and can be used to support globally important conservation **actions** (Holmes et al. 2019) and the continued conservation gains from rodent eradications on islands (Jones et al. 2016).

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Conflict of Interest/Competing interests

All authors, except for G. Baxter and F. Cunninghame, declare that they are working on or are likely to work on insular rodent eradication projects using anticoagulants.

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Not applicable

Consent to participate

Not applicable

Consent for publication

Not applicable

Availability of data and material

The data generated or analyzed during this study is included in this published article and its supplementary information files. Additionally, the database compiling the literature search findings and authors personal accounts is available from the corresponding author on request.

Code availability

Not applicable

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Figures

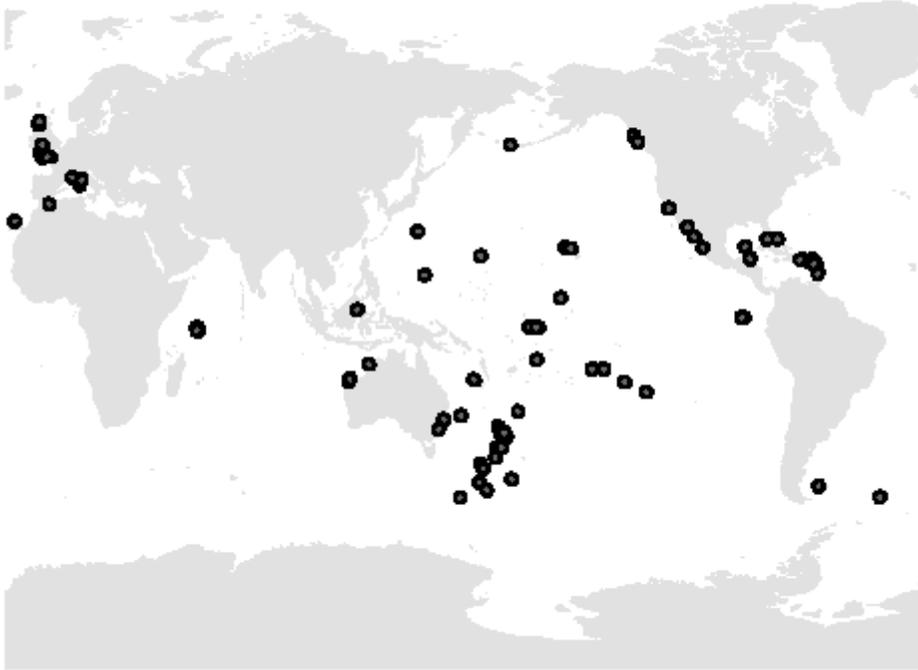


Figure 1

Location of islands (n=178) identified through this review where non-target wildlife mitigation actions were implemented during 82 rodent eradication attempts

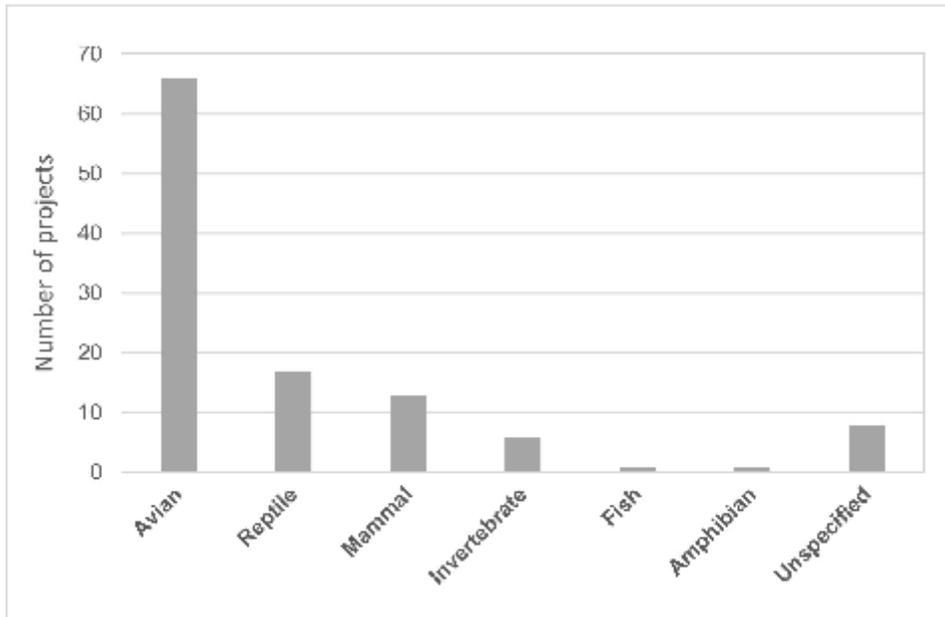


Figure 2

Non-target wildlife classes for which mitigation tactics were implemented during rodent eradication projects (n=82) on 178 islands, reported between 1990 and 2020

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