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Island prioritization for invasive rodent eradications with an emphasis on reinvasion risk

D. B. Harris · S. D. Gregory · L. S. Bull · F. Courchamp

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Abstract Invasive rodents occur on over 80% of the world's island groups, invasions are continuing, and rodent impacts on insular wildlife have been well demonstrated. The extent of this problem calls for tools to aid large-scale prioritisation among the many candidate eradication operations. As conservation funds are limited, biologists have responded with prioritisation systems based on financial cost-effectiveness. Instead, we claim that long-term conservation gain should be the primary focus when prioritising islands for invasive rodent eradication. This concept is embodied mainly by invasive rodent reinvasion risk, which we categorise as natural or anthropogenic, based on the mechanism of reinvasion and our ability to mitigate the risk. The result is a first-pass triage system that prioritises eradication programmes by their long-term conservation potential, not their immediate value for money. To construct a prioritization list, we group islands into units for simultaneous eradication, to minimize inter-island reinvasion risk, and then assign weights to levels of unit reinvasion risk and unit conservation value. The choice of parameter weights may depend on capacity for biosecurity action (i.e. reduction in reinvasion risk) and a choice of tertiary filter variables can further discriminate within priority ranks. We illustrate our prioritization framework with a case study on rodents in New Caledonia but explain how our system can be adapted to suit any invasive rodent species or island configuration.

Keywords Biosecurity · Conservation prioritization · Eradication · Invasive alien species · Reinvasion risk · Rodent

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D. B. Harris · S. D. Gregory · L. S. Bull · F. Courchamp
Ecologie, Systématique et Evolution, UMR-CNRS 8079, Université Paris-Sud, 91405 Orsay CEDEX, France

Present Address:
D. B. Harris (✉) · S. D. Gregory
Global Ecology Group, School of Earth and Environmental Sciences, University of Adelaide, Mawson Building, North Terrace, Adelaide, SA 5005, Australia
e-mail: donnabharris@googlemail.com

Present Address:
L. S. Bull
Boffa Miskell Ltd, PO Box 11340, Wellington, New Zealand

Introduction

Invasive rodents are among the greatest threats to island biodiversity, largely due to their impressive dispersal capabilities coupled with their heavy impacts on a variety of native fauna and flora. Over 80% of the world's oceanic island groups now host one or more species of invasive rodent and rodent invasions (and reinvasions) are continuing (Russell et al. 2008a). The black rat (*Rattus rattus*), brown rat (*Rattus norvegicus*), Pacific rat (*Rattus exulans*) and house mouse (*Mus musculus*) are particularly successful invaders that pose a significant global threat to insular biota (reviews include Courchamp et al. 2003; Harris 2009; Jones et al. 2008; Towns et al. 2006). In response to this problem, the number of invasive rodent eradication attempts, size of islands targeted and technological efficiency has been steadily increasing (Howald et al. 2007; Towns and Broome 2003).

Islands are, however, abundant and invasive species management funds scarce, and financial cost-effectiveness is increasingly considered a variable of predominant importance in conservation prioritisation (Brooke et al. 2007; Capizzi et al. 2010; Moore et al. 2010; Myers et al. 2000; Naidoo et al. 2006; Pimentel et al. 2005). We are of the opinion, however, that the *initial* prioritization of island eradications should not depend on immediate financial cost-effectiveness but on their potential contribution to long-term or sustainable wildlife conservation. A focus on financial cost-effectiveness should be reserved for the subsequent eradication planning stage after the required detailed information e.g. island inventory and eradication feasibility assessment have been collected/completed (Myers et al. 2000).

Multiple factors may jeopardize the long-term success of an eradication operation. Negative indirect (e.g. mesopredator release) or direct (e.g. non-target poisoning) effects of the eradication attempt itself can be mitigated by appropriate design of eradication strategy (e.g. Caut et al. 2009; Empson and Miskelly 1999; Olivera et al. 2010) but arguably the greatest threat is post-eradication reinvasion risk. Invasive rodents readily hitchhike aboard boats and ships (Moors et al. 1992; Russell et al. 2008a) but may also disperse naturally by swimming between islands. While *R. exulans* is unlikely to cover more than 50 m, *R. rattus* have crossed channels up to 750 m wide and *M. musculus* have crossed channels of approximately 500 m. *R. norvegicus*, however, is the superior swimmer: distributional data and recorded reinvasions suggest that they can cross up to 1 km of water and more rarely, under suitable conditions, up to 2 km (Burbidge 2004; Innes 2005; Russell and Clout 2005; Russell et al. 2005; Russell et al. 2008a; Towns et al. 2006). We incorporate reinvasion risk as the primary factor in our prioritization system and illustrate our framework with a case study of invasive rodent eradication prioritization in New Caledonia.

The archipelago of New Caledonia is a top priority for rodent eradication because of its high endemicity in flora (nearly 90%) and fauna (e.g. 83.5% in lizards) and the threats posed by anthropogenic impacts including mining, fire and introduced species (Beauvais et al. 2006; Gargominy 2003; Pascal et al. 2008). The creation of rat-free islands within the archipelago would therefore offer both immediate protection to the resident species and potential sites for future translocations as a safeguard against extinction.

We present each component of our framework and explain how a series of decisions can culminate in an island eradication prioritisation list that would direct eradication planning to maximise long-term wildlife conservation potential.

Materials and methods

Components of the framework

Reinvasion risk

The most obvious mechanism of rodent reinvasion risk to newly eradicated islands is natural dispersal between islands by swimming. This risk can be reduced by planning eradications in a meta-population context (Abdelkrim et al. 2005; Robertson and Gemmell 2004). This can be achieved by ascertaining the maximum natural dispersal distance of the target invasive species (e.g. through a literature review) and then grouping all adjacent, eradicable islands that are separated by less than this distance into eradication units. Once the islands of an eradication unit are simultaneously eradicated, as our framework recommends, the unit should then be immune to reinvasion by naturally dispersing individuals because it is surrounded by a buffer of ocean that exceeds the maximum natural dispersal distance of the target invasive species. Eradication units in most archipelagos will however be vulnerable to Anthropogenic Reinvasion Risk (ARR). This type of reinvasion describes human-mediated transport of invaders to islands. For example, there are records of rats swimming from moored boats or shipwrecks to islands and escaping from cargo landed on islands (Russell et al. 2008a). An eradication unit's ARR score (e.g. low, med, high) is therefore based on the frequency of human visitation, accessibility or similar factors.

In its most general form, our framework considers ARR the only reinvasion risk to eradication units. In reality, however, many archipelagos contain at least one ineradicable island or are within the target invasive species' dispersal distance from a continental coastline. Larger, heavily populated islands with international traffic and obviously continents themselves are not suitable for invasive rodent eradication and therefore exist as (ineradicable) sources of naturally dispersing rodents. To admit this common situation as a special case of our framework, we define Natural Reinvasion Risk (NRR). Where the closest source island/continental coast is further than the natural dispersal distance of the invasive species, an eradication unit's NRR score will be low. Where the source is situated within natural dispersal distance of the eradication unit, the unit NRR score (e.g. med or high) will be proportional to the distance of that unit from the source island/coast or based on known swimming capabilities of the target species (Fig. 1). In theory, NRR is never zero for any eradication unit because individuals may travel beyond their maximum natural dispersal distance by 'assisted' natural dispersal—e.g. by rafting on a vegetation mat (e.g. Thornton 1971).

Conservation value

Conservation value (CV) is a measure of the conservation gain in the event that an eradication unit is successfully eradicated of the target invasive species. Accordingly, CV will be best characterized using species that are known to be negatively impacted by the target invasive species. Indices of biodiversity or endemicity are commonly used in conservation prioritisation exercises (see Brooks et al. (2006) for a review of approaches) and species vulnerability to the target invasive might be usefully incorporated in eradication prioritisation exercises (e.g. Brooke et al. 2007; Brown 2001). CV scores might derive solely from currently resident fauna and flora or incorporate the future conservation value of islands following a planned species translocation or island restoration program (e.g. Brown 2001). The focal taxa and the quantification of CV will depend in part on local scientific and management consensus and on data availability. A major advantage of our system is that

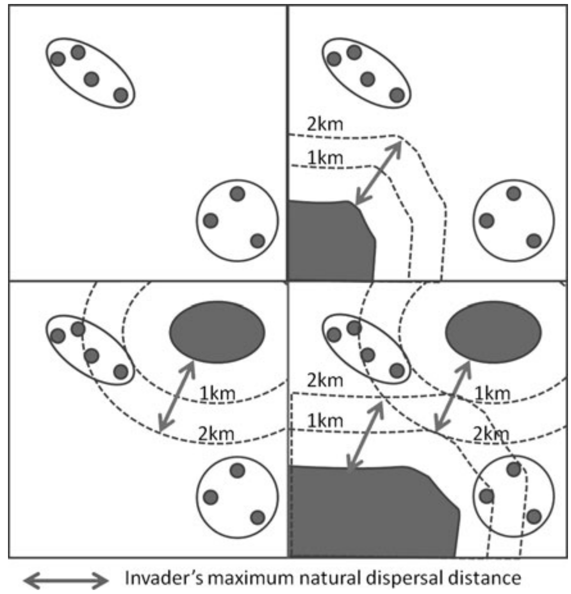

Fig. 1 The *upper left* schematic illustrates the construction of eradication units where islands are grouped if within the invaders maximum natural dispersal distance (2 km in this example) of each other. The *upper right box* illustrates the simplest form of our framework where eradication units lie further than the invaders maximum natural dispersal distance of an ineradicable source island/continent. The *lower boxes* illustrate the special case of our framework where an eradication unit lies within the invaders maximum natural dispersal distance of an ineradicable source island/continent. In the *lower right box* the upper eradication unit might be given a med NRR score and the lower unit a high NRR score according to the distance of the closest island in the eradication unit to the ineradicable source

crude scale data is sufficient for a first-pass triage system to identify priority islands, with long-term conservation potential, for rodent eradication.

Constructing prioritization scenarios

Ultimately, our framework is designed to provide a list of eradication units for invasive rodent eradication ranked by their potential for long-term wildlife conservation. Such ranking schemes give no quantitative indication of how much 'more important' is a high than a low rank eradication unit (Murdoch et al. 2007) but quantitative evaluation of the benefit versus cost of eradicating an invasive species is notoriously difficult without detailed island-specific information (Donlan and Wilcox 2007; Myers et al. 1998). Consequently, and to extend its use to include qualitative variables, we chose to use a simple weighted ordinal scoring system.

Each variable's scores must be assigned a weighting scheme representing both the relative importance of variables to achieving long-term conservation in the archipelago and the relative importance of levels within variables. Such weighting schemes should be chosen to represent feasible management scenarios. For example, a baseline scenario might be represented by weights assigned to the reinvasion risk (NRR and ARR) scores that exceed those assigned to CV scores. This would reflect the view that long-term conservation potential is threatened by reinvasion risk so that CV should order islands only within reinvasion risk levels. As the relative contribution of natural versus human-mediated dispersal to island reinvasion rates is largely unknown, identical weight sets may be applied to both NRR and ARR levels in a default or baseline scenario.

The establishment of island biosecurity systems, designed to prevent and detect alien species incursions, may represent a special case of the baseline scenario in which ARR reinvasion risk levels are repressed (Broome 2007; Jay et al. 2003). For rodents, biosecurity mechanisms to limit ARR include compulsory pre-departure vessel inspections, on-vessel control devices, public information campaigns and guidelines and mooring regulations (Moors et al. 1992). Preventing the departure and intercepting the arrival of swimming rodents (i.e. NRR) is, however, arguably more difficult as invasion sources and entry points are more diffuse, i.e. entire coasts rather than single ports (Russell et al. 2009, 2010). Source populations may be controlled to limit the probability of departure and on-island devices can be maintained to detect and control new arrivals but such techniques are labour intensive on large or multiple islands in the long-term (Broome 2007; Russell et al. 2008b). Due to these perceived difficulties we consider NRR levels fixed in our framework while ARR levels can be reduced under a biosecurity scenario. It has been demonstrated, for rodents, that where permanent surveillance systems employ multiple biosecurity devices with full island coverage and contingency responses employ alternative methods, success rates in detecting and removing reinvaders is very high (Russell et al. 2008b). All levels of ARR could therefore be assigned the same weight to effectively drop ARR from the framework, simulating the successful implementation of biosecurity as a management strategy (e.g. Moore et al. 2010). More conservatively, a manager may prefer to vary weights between ARR levels to reflect a predicted or suspected correlation between ARR and visitation rate, despite the establishment of biosecurity.

For a given scenario (e.g. baseline or biosecurity) the weighted scores for NRR, ARR and CV are then summed for each eradication unit to give a priority score which is then used to order the units by priority for eradication.

Case study: invasive rodents in New Caledonia

The first step of the prioritization process was to identify all New Caledonian islands. The archipelago has 5 main populated islands: Grande Terre, Isle of Pines and the three main Loyalty Islands (Ouvéa, Lifou and Maré) and we mapped and labeled another 560 islands, islets, rocks and rock clusters using Google Earth. If an island, islet or rock was $\leq$20 m from an adjacent island, we considered it part of the larger island. Although island names and locations were derived mostly from the Geonames database (www.geonames.org), inspection of satellite imagery on Google Earth revealed 213 unnamed land masses. Sand cays and small rocks were included because rodents can survive on them when seabirds are present and they may serve as 'stepping-stones' to other islands.

The distribution of invasive rodents in New Caledonia is not known. Larger islands with human populations host various combinations of the four rodent species (*R. rattus*, *R. norvegicus*, *R. exulans* and *M. musculus*) but their presence/absence on most other islands has not been documented and it is especially difficult to confirm rodent absence. Consequently, we adopted the precautionary approach of assuming the presence of at least one species on every island until islands are surveyed and this is proved or disproved. In effect then, the final list of islands will be a priority list for invasive rodent surveys (and other wildlife surveys), followed by eradication where needed.

Past international success dictates that aerial baiting could be considered on islands of similar size to the largest eradicated to date: 11,300 ha Campbell Island, New Zealand (Howald et al. 2007). However, unlike uninhabited Campbell, the similar sized New Caledonian islands of Ouvéa and Pines host large human populations, so that livestock, pets and children would be at risk of poisoning from aerial broadcast bait.

Although bait stations could be used to minimize non-target poisoning risk on heavily-populated islands (Grande Terre, Ouvéa, Lifou, Maré and Pines), they are all much larger than the largest island eradicated to date using bait stations (3,105 ha Langara Island, British Columbia, Canada; Howald et al. 2007). Ground-based operations are more sensitive to logistic coordination (K. Broome pers. comm.) and operations of this spatial scale are therefore unlikely to succeed. In addition, islands with large human populations incur high rodent reinvasion risk via domestic traffic and cargo (Oppel et al. 2011). We therefore classified Grande Terre, Ouvéa, Lifou, Maré and Pines as ineradicable and considered them permanent sources of invasive rodents.

Eradicable islands spaced less than 2 km apart were grouped into eradication units that should be considered for simultaneous rodent eradication to minimize inter-island reinvasion risk (Robertson and Gemmell 2004). Because we are uncertain of its distribution in the archipelago but know that *R. norvegicus* at least occurs in the Loyalties (on Lifou) and on Grande Terre, and swims further (up to 2 km) than the other invasive rodents in New Caledonia, we took the precautionary principle and used 2 km as the maximum natural dispersal distance of rodents in the archipelago. Classifying the eradicable 560 New Caledonian islands into eradication units using this criterion resulted in 240 eradication units.

Each individual island was assigned an ARR score based on the following criteria: Permanently inhabited islands or those with organized tourism were allocated a HIGH ARR. Islands with no permanent inhabitation or organized tourism but evidence of recreational (private) visits were allocated a MED ARR. This was also considered the default ARR score because HIGH and LOW ARR islands are easy to identify from, for example, census data and reserve access bans, respectively. Uninhabited islands that were either geographically isolated (e.g. Chesterfield islands), had difficult landing (e.g. Matthew) or an access ban (e.g. Kié Island of the Réserve Marine Intégrale Yves Merlet) were allocated a LOW ARR. Each eradication unit was then allocated an ARR score according to the highest ARR score of its constituent islands.

Data sources used for the quantification of ARR included government census data from 2004 (www.isee.nc) and the online Geonames database (www.geonames.org: check for "populated places"). Reserve details were also found online (www.ecologie.gouv.fr). Due to the paucity of published data at the level of the individual island, we conducted standardized Google searches (www.google.com) for every island. We checked only the first page of results and relevant links and documents were perused for information on visitation and tourism. Personal blogs were included as evidence of visitation. Google Earth satellite imagery (and attached photographs) was inspected for every island in an attempt to identify whether buildings, wharfs or airstrips were present and whether there were other signs of visitation (e.g. moored boats, agriculture).

New Caledonia represents a special case of our framework where ineradicable source islands are present. Consequently, we calculated NRR for all eradication units. All eradication units >2 km from an ineradicable island were allocated a LOW NRR score because rodents can't swim this far. If the unit is 1-2 km from an ineradicable island then the unit was allocated a MED NRR score because rodents may occasionally swim this distance, and if <1 km, the unit was allocated a HIGH NRR score because rodents may routinely swim this distance (Fig. 1). Eradication unit distance to rodent source island was estimated using the distance tool in Google Earth and was measured from the island in the eradication unit that was closest to an ineradicable source island (http://earth.google.com).

New Caledonia has been the focus of a recent survey to identify global hotspots for bird conservation known as Important Bird Areas (Birdlife International 2008; Spaggiari et al. 2007). Of the 32 IBAs in New Caledonia, 9 of these either are (4), or contain (5), the eradication units designated in our study (Fig. 2). The other IBAs lie on ineradicable islands (mostly on Grande Terre). It is well established that seabirds are vulnerable to rodent predation (Jones et al. 2008) so we used the IBA survey data to rank the IBAs by seabird conservation value (Online Resource 1). We initially ranked the nine IBAs by the number of endemic species present. As endemicity alone does not indicate global importance of a particular region, we then ranked IBAs by global importance by summing the percentage of breeding pairs of the global total breeding pairs for each species across all breeding seabird species within each IBA (see data in Online Resource 1). For example, although the Poindimié IBA hosts no endemic breeding species it hosts 27.3%

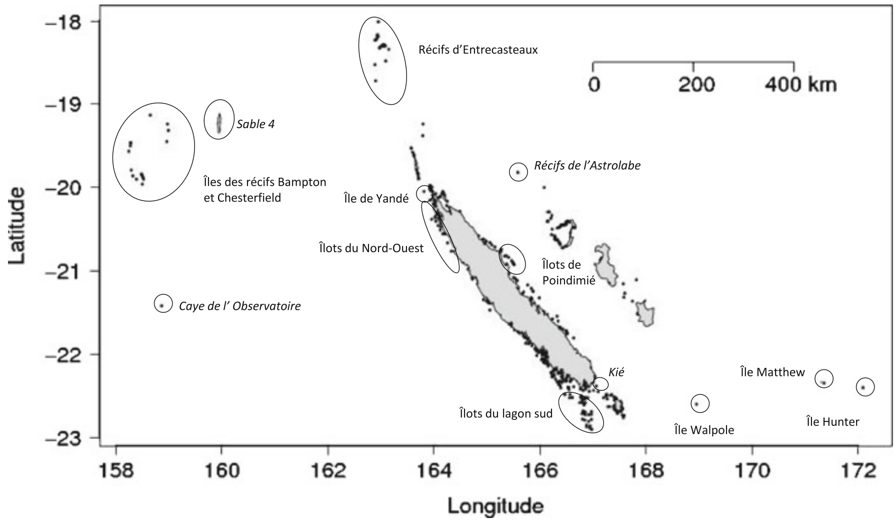

Fig. 2 Map of New Caledonia showing the Important Bird Areas (Birdlife International 2008; Spaggiari et al. 2007) that contain rodent-eradicable islands, plus some additional non-IBA islands (names italicized) that feature in the top 50 eradication units table (Table 2)

Table 1 Final ranks for IBAs derived from the average of endemicity rank and importance rank

IBA name	No. endemic sub-species[a]	Endemicity rank	Importance score	Importance rank	Average rank	Final rank
Lagon Sud	3	5	46.107	9	7	7
Nord-Ouest	3	5	8.475	6	5.5	6
Bamp-Chest	1	3	9.254	7	5	5
Poindimié	0	1	27.709	8	4.5	4
D'Entrecast	0	1	6.313	5	3	3
Île Hunter	0	1	4.255	4	2.5	2
Île Walpole	0	1	3.927	3	2	1
Île Matthew	0.5	2	1.89	2	2	1
Île de Yandé	1.5	4	0	1	2.5	2

[a] Endemic sub-species: definitely breeding – 1 point, probably breeding – 0.5 points

of the global population of *Sterna dougalli bangsi* which gives this IBA the second highest rank for importance (27.3% + 0.001% (5/500,000 *Anous stolidus pileatus*) + 0.0006% (3/500,000 *Sterna bergii cristata*) + 0.4% (400/100,000 *Sterna sumatrana*)). This additive treatment means that higher species diversity may also increase the importance score, all other things being equal. The average rank was then calculated for each IBA ((endemicity rank + importance rank)/2) (Table 1). Final ranks (Table 1) were then assigned to the eradication units falling geographically within each IBA. The higher the rank, the higher the Conservation Value. This zonal approach is necessary when prioritising for seabird conservation because seabirds breed on different islands in an area, in different years. Despite the archipelago-level identification of IBAs, data remain incomplete for many islets outside these zones (Baudat-Franceschi pers. comm.) and non-IBA islands that remain to be surveyed for seabirds were given a rank of 0 as it is impossible to prioritise for missing conservation value.

We used our framework to prioritize New Caledonian eradication units under two scenarios: (1) a 'baseline' scenario and (2) a 'biosecurity' scenario. As there is no a priori reason to assume otherwise, the weight sets for NRR and ARR are equal with low-medium–high risk score weights of 30:20:10 respectively for NRR and ARR. These weights are arbitrary and were chosen because they allow CV (score range 0–7—i.e. <10) to act as a secondary filter after Reinvasion Risk score (NRR + ARR) has prioritized the eradication units. For example, although both Carrey and Caye Skeleton (Table 2) have a low risk of natural reinvasion by swimming rodents (NRR of 30) and a low risk of reinvasion via human-mediated dispersal (ARR of 30), Nord Ouest IBA islands have a higher CV score than the Bampton and Chesterfield IBA islands (6 vs. 5; Table 1) so that the final summed scores are 66 and 65 for Carrey and Caye Skeleton respectively. In other words, once eradication units are sorted by reinvasion risk, ranks are further separated by conservation value. In the biosecurity scenario ARR effectively drops out of the framework because all ARR scores are set to an equal weight. The weight sets in our example are again 30:20:10 for NRR but 1:1:1 for ARR. This simulates the effective implementation of biosecurity on all islands, negating anthropogenic reinvasion risk.

Results

Baseline scenario

The priority list for New Caledonian rodent eradications generated from the baseline scenario obviously favors eradication of units least likely to be reinvaded by invasive rodents (Table 2). Where information on CV is available it has served to rearrange the list within reinvasion risk levels so that the eradication units at the top of the list would have the greatest long-term conservation potential based on current knowledge and assuming limited biosecurity. Under the baseline scenario, the top 50 eradication units include Carrey island in the Nord Ouest IBA, ranked highest as a result of its relatively isolated location coupled with relatively high seabird conservation value. The extremely isolated Chesterfield islands then dominate followed by Hunter, Matthew and Walpole islands which are not only isolated but have difficult landings, contributing to their low visitation rates. These eradication units are followed by units which remain to be surveyed for their seabird composition and these should be viewed as priorities for inventory work, followed by rodent eradication if required (Table 2; Fig. 2).

Biosecurity scenario

If an efficient biosecurity program were implemented, then the priority list for rodent eradication differs substantially from that generated under the baseline scenario (Table 2). This is because the priority list generated from the biosecurity scenario prioritises eradication units with high CV which had been down-weighted to a lower rank under the baseline scenario because of their high ARR. In other words, if ARR can be reduced through effective biosecurity on islands then managers can focus more on prioritizing islands by their conservation value, especially where NRR is also low. This explains why all eradication units in Lagon Sud assemble to form the top 23 priorities under the biosecurity scenario (Table 2). They all have medium or high ARR, which we assume can be minimised by biosecurity measures, and they all have low NRR, being more than 2 km from their nearest ineradicable source of rodents (Fig. 2).

Discussion

Eradication prioritization framework: a conservation planning tool

In this article we have presented a general framework for prioritizing invasive rodent eradications on islands that emphasises reinvasion risk and should direct eradication planning, including cost-effectiveness analysis, to those islands with the highest long-term conservation potential. Our approach reflects our view that prioritizing islands for invasive species eradication should no longer be focused solely on getting the biggest immediate “bang for the buck” but, rather, on the potential for long-term wildlife conservation. In our view, financial costs should be omitted from the initial prioritization phase and instead reserved as an important quantitative variable in the later planning stage.

At its core, our framework adopts a meta-population approach to insular invasive species by focusing on groups of islands, or eradication units, rather than single islands. In the development of this framework we also advanced the concept that reinvasion risk, the ultimate threat to eradication success, can be broken down into two types based on the mechanism of reinvasion and our ability to mitigate the risk: (1) Anthropogenic reinvasion risk or human-facilitated transfer which we can control through preventative biosecurity and (2) Natural reinvasion risk defined by natural dispersal of invaders from ineradicable source islands/mainland over which long-term, large-scale

Table 2 List of the top 50 eradication units ranked in priority order according to the baseline and biosecurity scenarios for New Caledonia. Low, med and high NRR and ARR scores are given weightings of 30, 20 and 10, respectively under the baseline scenario and ARR score weights are all reduced to 1 for the biosecurity scenario

Baseline							Biosecurity						
Island(s) in unit	No. islands	IBA name	NRR	ARR	CV	Total	Island(s) in unit	No. islands	IBA name	NRR	ARR	CV	Total
Carrey	1	Nord-Ouest	30	30	6	66	Uaterembi	1	Lagon Sud	30	1	7	38
Caye Skeleton	1	Chesterfields	30	30	5	65	Gi	1	Lagon Sud	30	1	7	38
Avon 2	1	Chesterfields	30	30	5	65	Rédika	1	Lagon Sud	30	1	7	38
Avon 3	1	Chesterfields	30	30	5	65	Koko	1	Lagon Sud	30	1	7	38
Îlot du Passage	1	Chesterfields	30	30	5	65	Nge	1	Lagon Sud	30	1	7	38
Îlot de Sable	1	Chesterfields	30	30	5	65	Uié	1	Lagon Sud	30	1	7	38
Avon 1	1	Chesterfields	30	30	5	65	Iéroué	1	Lagon Sud	30	1	7	38
Renard	1	Chesterfields	30	30	5	65	Totéa	1	Lagon Sud	30	1	7	38
Bampton	1	Chesterfields	30	30	5	65	Teré	1	Lagon Sud	30	1	7	38
Southeast Bampton Reef (island)	1	Chesterfields	30	30	5	65	Uua	1	Lagon Sud	30	1	7	38
Avon 4	1	Chesterfields	30	30	5	65	Atire	1	Lagon Sud	30	1	7	38
Les Trois Îlots du Mouillage (×3), NC16, NC17	5	Chesterfields	30	30	5	65	Mboré	1	Lagon Sud	30	1	7	38
Hunter	1	Hunter	30	30	2	62	Ndo	1	Lagon Sud	30	1	7	38
Matthew	1	Matthew	30	30	1	61	NC163	1	Lagon Sud	30	1	7	38
Walpole	1	Walpole	30	30	1	61	NC183	1	Lagon Sud	30	1	7	38
Kié	1		30	30	0	60	Caye Mambaé	1	Lagon Sud	30	1	7	38
Caye de l' Observatoire	1		30	30	0	60	Ndi	1	Lagon Sud	30	1	7	38
Sable 4	1		30	30	0	60	Mato	1	Lagon Sud	30	1	7	38
Récifs de l' Astrolabe	>1		30	30	0	60	NC97	1	Lagon Sud	30	1	7	38
Uaterembi	1	Lagon Sud	30	20	7	57	Kouaré	1	Lagon Sud	30	1	7	38
Gi	1	Lagon Sud	30	20	7	57	Uatio, Ua	2	Lagon Sud	30	1	7	38
Rédika	1	Lagon Sud	30	20	7	57	Nda, NC114	2	Lagon Sud	30	1	7	38
Koko	1	Lagon Sud	30	20	7	57	Uo, NC213, NC214, NC215, NC216	5	Lagon Sud	30	1	7	38
Nge	1	Lagon Sud	30	20	7	57	Carrey	1	Nord-Ouest	30	1	6	37
Uié	1	Lagon Sud	30	20	7	57	Tiam' bouène	1	Nord-Ouest	30	1	6	37
Iéroué	1	Lagon Sud	30	20	7	57	Double	1	Nord-Ouest	30	1	6	37
Totéa	1	Lagon Sud	30	20	7	57	Yan'dagouet	1	Nord-Ouest	30	1	6	37
Teré	1	Lagon Sud	30	20	7	57	Pouh	1	Nord-Ouest	30	1	6	37
Uua	1	Lagon Sud	30	20	7	57	Deverd	1	Nord-Ouest	30	1	6	37
Atire	1	Lagon Sud	30	20	7	57	Kendec	1	Nord-Ouest	30	1	6	37
Mboré	1	Lagon Sud	30	20	7	57	Table and NC14	2	Nord-Ouest	30	1	6	37
Ndo	1	Lagon Sud	30	20	7	57	Ouanne, NC181	2	Nord-Ouest	30	1	6	37
NC163	1	Lagon Sud	30	20	7	57	Caye Skeleton	1	Chesterfields	30	1	5	36
NC183	1	Lagon Sud	30	20	7	57	Avon 2	1	Chesterfields	30	1	5	36
Caye Mambaé	1	Lagon Sud	30	20	7	57	Avon 3	1	Chesterfields	30	1	5	36
Ndi	1	Lagon Sud	30	20	7	57	Îlot du Passage	1	Chesterfields	30	1	5	36
NC97	1	Lagon Sud	30	20	7	57	Îlot de Sable	1	Chesterfields	30	1	5	36

Table 2 continued

Baseline							Biosecurity						
Island(s) in unit	No. islands	IBA name	NRR	ARR	CV	Total	Island(s) in unit	No. islands	IBA name	NRR	ARR	CV	Total
Nda, NC114	2	Lagon Sud	30	20	7	57	Avon 1	1	Chesterfields	30	1	5	36
Uo, NC213, NC214, NC215, NC216	5	Lagon Sud	30	20	7	57	Renard	1	Chesterfields	30	1	5	36
Tiam' bouène	1	Nord-Ouest	30	20	6	56	Bampton	1	Chesterfields	30	1	5	36
Double	1	Nord-Ouest	30	20	6	56	Southeast Bampton Reef (island)	1	Chesterfields	30	1	5	36
Yan'dagouet	1	Nord-Ouest	30	20	6	56	Avon 4	1	Chesterfields	30	1	5	36
Pouh	1	Nord-Ouest	30	20	6	56	Loop	1	Chesterfields	30	1	5	36
Table and NC14	2	Nord-Ouest	30	20	6	56	Longue	1	Chesterfields	30	1	5	36
Ouanne, NC181	2	Nord-Ouest	30	20	6	56	Les Trois Îlots du Mouillage (×3), NC16, NC17	5	Chesterfields	30	1	5	36
Loop	1	Chesterfields	30	20	5	55	Solitaire (=Faux Tabac)	1	Poindimié	30	1	4	35
Longue	1	Chesterfields	30	20	5	55	Banc de sable 1	1	Poindimié	30	1	4	35
Solitaire (= Faux Tabac)	1	Poindimié	30	20	4	54	Pètèmen	1	Poindimié	30	1	4	35
Banc de sable 1	1	Poindimié	30	20	4	54	Hwayé Bwadoa	1	Poindimié	30	1	4	35
Pètèmen	1	Poindimié	30	20	4	54	Bayes, Bois de Fer, Banc de sable 2, Banc de sable 3	4	Poindimié	30	1	4	35

The CV score varies between 0 and 7. Total score is the sum of ARR, NRR and CV scores. NC codes refer to unnamed islands

control is more difficult. Our framework ranks eradication units primarily by these types of reinvasion risk which allows the manager to assess each eradication unit's long-term conservation potential. Our framework can also be used to construct and compare prioritization lists under candidate management strategies.

We illustrated the utility of this tool by prioritizing New Caledonian islands for invasive rodent eradication and comparing baseline and biosecurity management scenarios. The results suggest that implementing biosecurity could increase the rate of seabird conservation achieved as eradication units with higher conservation value are prioritized regardless of their ARR status (in this case the Lagon Sud islands are favoured). Encouragingly, where surveillance systems and contingency responses are properly maintained, biosecurity can be 100% effective. For example, rat incursions are annual events on many islands in New Zealand but bait stations and traps have successfully prevented reinvasion (Russell et al. 2008a). Nevertheless, detection of rats can be extremely challenging (e.g. Thorsen et al. 2000, Russell et al. 2005), biosecurity is expensive in perpetuity (e.g. surveillance on Motuhoropapa Island costs NZ$740 per visit, with an additional NZ$1420 per contingency response (Russell et al. 2008b) and despite best efforts, is always fallible (e.g. after 13 years of rat-free status a breeding population of *R. norvegicus* has recently established on Ulva Island; DOC 2011). Clearly, the benefits in terms of long-term conservation value would need to justify the long-term logistic and financial costs particularly where eradication on a low reinvasion risk island is a contending project in the prioritisation process.

Interestingly, the first documented rodent eradication campaign in New Caledonia focused on the Lagon Sud (Bell 1998). The multi-island operation was apparently successful but there is a need to check these islands for reinvasion (Spaggiari et al. 2007).

More recently, the focus has been on restoration within the Nord-Ouest Lagon IBA where three islands (Tiam' bouène, Double and Table) were baited in 2007 and recently confirmed rat-free (SCO 2011). These islands are ranked after the Lagon Sud islands under the biosecurity scenario (Table 2). Indeed, a biosecurity plan has been developed by the North Province authority and the Société Calédonienne d'Ornithologie with an emphasis on making recreational users aware of the rat-free status of islands and aware of practices for preventing incursions (Birdlife International 2009). Once the ongoing success of the Lagon Sud eradications has been verified it should also be a priority to establish a biosecurity plan in the south. Such a plan might include compulsory boat inspections and camping protocols as enforced, for example, in the Galápagos Islands (Parque Nacional Galápagos 2008).

The implementation of a biosecurity strategy may not, however, always be optimal. This would be especially pertinent in archipelagos where ARR and CV are negatively correlated, for example where there is a long history of anthropogenic visitation and consequent exploitation or habitat destruction (i.e. high ARR and low CV). In this case, the conversion of high and medium ARR to low levels through biosecurity would not cause the affected eradication units to rise to the top of the priority list because their CV scores are low (possibly as a consequence of human impact). At the other extreme, archipelagos with very low levels of human residence and visitation across all constituent islands would also benefit less from the implementation of an expensive biosecurity campaign and funds might be better channelled into eradication of rodents that may have arrived during historical expeditions or ship wrecks. Nevertheless, routine checks for reinvasion are necessary especially where conservation value is disproportionately high (e.g. the subantarctic islands of New Zealand).

Alternatively, where biosecurity is desirable but the capacity to enforce it in perpetuity is doubtful, some conservation managers may prefer to opt for the long-term conservation/baseline strategy in which eradication units are prioritized primarily on the basis of reinvasion risk and secondarily by conservation value. Initially, cost-effectiveness (conservation value/financial cost) might appear relatively low, but pre-eradication survey may reveal important species, good seabird colonization/re-colonization potential or habitat suitable for translocation of rodent-vulnerable species. Indeed, colonization or translocation potential could be incorporated as additional secondary factors, or integrated with CV, in our framework. The Chesterfield Islands, Hunter, Matthew and Walpole (dominating the top 20 under the baseline scenario) have been the focus of recent survey and restoration assessments but the sites are very remote as well as being logistically and physically difficult to access, requiring multi day charters (Birdlife International 2009). Naturally, such difficulties are also reflected in eradication costing. We argue, however, that the long term benefits of eradications on remote, low reinvasion risk islands could well outweigh the initial financial costs and logistic difficulties. For example; within 7 years of rat eradication on subantarctic Campbell Island, an undescribed species of snipe (*Coenocorypha sp.*) recolonised, white-chinned petrels (*Procellaria aequinoctialis*) and grey-backed storm petrels (*Oceanites nereis*) returned to breed, and an endemic teal species (*Anas aucklandica nesiotis*) was re-estabished (Bellingham et al. 2010).

Our prioritized list should, in the first instance, be used to target the collection of island-specific data. In addition to establishing the presence of the target invasive and the species that may benefit by its eradication a preliminary island visit should also aim to assess the likelihood of undesirable negative impacts of the eradication e.g. primary or secondary poisoning (Brown 2001; Empson and Miskelly 1999) or potentially damaging higher level trophic interactions (Bull and Courchamp 2009; Caut et al. 2009; Courchamp et al. 2003; Harris and Macdonald 2007). Monitoring and mitigation of such impacts can complicate and significantly increase the cost of an eradication campaign. This is illustrated by comparison of the USD 1.7 m budget for 11,300 ha Campbell Island where little monitoring/mitigation was necessary with USD 1.8 m for 296 ha Anacapa Island where extensive monitoring/mitigation was necessary (Roberts 1999, Howald et al. 2010). An understanding of the ecology as well as the practicalities of species eradication (e.g. accessibility, topographic complexity) on a specific island is therefore necessary to generate a realistic budget, in turn facilitating quantification of the relative cost-effectiveness of a particular eradication. These detailed costings should then be made available to the wider conservation community to facilitate future planning (Donlan and Wilcox 2007, Naidoo et al. 2006).

In practical terms, the outputs of our framework may be used as a structured, transparent tool to help raise funds for the preliminary island visits, field surveys and planning necessary to produce a detailed, cost-efficient eradication strategy for islands with the highest long-term conservation potential. For example, the baseline priority list for New Caledonia highlights the importance of raising funds for field surveys on Kié, Caye de l' Observatoire, Sable 4 and Récifs de l' Astrolabe before the relative cost-effectiveness of their eradication can be ascertained. The islands eradicated to date in the Lagon Sud and Nord-Ouest Lagoon IBAs, while of high conservation priority, are ranked lower due to their relatively high reinvasion potential illustrating the need for continuing funding to support biosecurity campaigns in these regions.

Extending the framework

The flexibility of our prioritization framework makes it universally applicable to any island group or archipelago. Conservation value can be measured for regions or individual islands and based on continuous data or relative qualitative ranks and our framework can direct the collection of missing data. This is helpful as island inventory data is rarely comprehensive (Donlan and Wilcox 2008). Some island groups may not contain large, ineradicable islands and may be far from continental land (also an ineradicable source of invaders) in which case the eradication unit NRR may not be applicable. Instead just the unit ARR levels would be employed in the prioritization.

Although our framework has been designed for invasive rodents, it could be just as useful for other small mammal species with known dispersal distance (NRR), *accidental* anthropogenic transfer risk (ARR) and defined impacts on native species (essential for CV quantification) e.g. the Indian mongoose. By equalising weights among levels of superfluous factors the framework can also be applied to other species but the usefulness of the tool is then variable. For example, despite clear impacts on native species, larger mammals such as cats and goats reinvade islands only via *deliberate* anthropogenic transfer which is more difficult to predict i.e. visitation rate is not a sufficient proxy (e.g. Carrion et al. 2011). The system may be more usefully applied to birds, invertebrates and plants with estimable NRR (e.g. smooth-billed ani, coconut palm) or ARR (e.g. ants) but their impacts may be difficult to define (e.g. Vilà et al. 2011).

Both natural and anthropogenic dispersal propensity should be considered on a region by region basis as they can vary among populations and environmental conditions e.g. swimming distance may vary with ocean currents, sea temperature, etc. and likelihood of anthropogenic transport may vary with source population density (in turn a function of human activity or local food availability) (Russell et al. 2008a). Genetic assignment methods are being increasingly employed to discriminate between survivors and reinvaders when target pests are detected post-eradication. By revealing the source of reinvaders, these methods are providing information on relative rates of NRR to ARR and weightings can be adjusted accordingly within our framework (Russell et al. 2010).

Although our framework can function using coarse categorical data, such data performs poorly when trying to distinguish between eradication units with shared ranks in the prioritization list. Additional variables that could act as tertiary filters to separate equal ranks include island area, number of islands per unit and human population density in the closest settlement. Smaller islands tend to host simpler ecosystems, reducing the need for extensive or complex non-target mitigation action, the likelihood of having to deal with multiple invasive species and the management effort, logistics and capability required for the bait broadcast phase, all of which may jeopardize the success of an eradication campaign (K. Broome pers. comm.). Ranking eradication units by island size composition or largest island may therefore add a feasibility slant to the prioritization, as might the number of islands to be eradicated within each unit. Regardless of the nature and number of filters applied we reiterate that our system remains a first-pass, qualitative triage system designed to flag islands with the greatest potential for long-term conservation gain following alien species eradication. Our system is a precursor to finer scale eradication prioritisation, directing the island-specific data collation and survey work necessary for detailed planning and estimation of cost-effectiveness.

Conclusions

We present a framework to prioritize invasive species eradications from islands considered within a metapopulation context and with an emphasis on their reinvasion risk. Our framework is particularly well suited to invasive rodents because their impacts have been well documented and knowledge of their natural and commensal dispersal mechanisms facilitates quantification of reinvasion risk. However, the model parameters and their relative emphasis can be adjusted in various ways depending on the species to be eradicated, the species to be conserved, local geography, logistic capacity and project feasibility. Due to its modest initial data requirements, our approach should break any deadlock imposed by lack of island-specific biological data. Ultimately our system will promote cost-efficient eradication campaigns that ensure the long-term conservation of insular biodiversity.

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