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REVIEW PAPER

Deciding when to lend a helping hand: a decision-making framework for seabird island restoration

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Abstract Following the removal of an introduced species, island restoration can follow two general approaches: passive, where no further intervention occurs and the island is assumed to recover naturally, and; active, where recovery of key taxa (e.g. seabirds) is enhanced by manipulating movement and demography. Steps for deciding between these techniques are: (1) outlining an explicit restoration goal; (2) building a conceptual model of the system; (3) identifying the most effective management approach; and (4) implementing and monitoring outcomes. After decades of island restoration initiatives, retrospective analysis of species' responses to active and passive management approaches is now feasible. We summarize the advantages of incorporating these analyses of past restoration results as an initial step in the decision-making process. We illustrate this process using lessons learned from the restoration of seabird-driven island ecosystems after introduced vertebrate eradication in New Zealand. Throughout seven decades of successful vertebrate eradication projects, the goals of island restoration have shifted from passive to active enhancement of island communities, which are heavily dependent on burrow-nesting petrel

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population recovery. Using a comparative analysis of petrel response to past predator eradications we built a conceptual model of petrel recovery dynamics and defined key site and species characteristics for use in a stepwise decision tree to select between active or passive seabird population management. Active restoration techniques should be implemented when seabird populations are absent or declining; and on islands with no nearby source colony, small remnant colonies, highly altered habitat with shallow soil and slopes, and with competitive species pairs. As we continue to restore complex island communities, decision-making tools using a logical, step-wise framework informed by previous restoration successes and failures can aid in increasing understanding of ecosystem response.

Keywords Adaptive management · Decision tree · Burrowing seabirds · Eradication · New Zealand · Prioritization · Recovery

Introduction

Islands hold a disproportionately large percentage by area of global biodiversity and are increasingly important repositories for populations or ecosystems eliminated from the mainland (Daugherty et al. 1990; Mittermeier et al. 1998; Kier et al. 2009). Unfortunately, island ecosystems are also more susceptible to disturbance and have experienced high extinction rates—primarily due to the introduction of alien species (King 1985; Courchamp et al. 2003). Although advancements have been made in alien species eradications and ecosystem restoration, islands' remote locations and limited infrastructure make conservation efforts expensive (Donlan 2007; Helmstedt et al. 2016). Priority setting and decision-making tools are thus especially important in allocating limited resources while maximizing successful outcomes of island restoration.

Islands are particularly important for seabirds, typically providing terrestrial predator-free nesting habitat in proximity to pelagic feeding areas. Similarly, seabirds are particularly important members of island ecosystems, where their presence drives ecological processes (Mulder et al. 2011a). On islands where seabirds nest (henceforth "seabird islands"), seabird guano enriches the soil with marine-derived nutrients and nest-building disturbs vegetation and aerates the soil (Mulder et al. 2011b; Smith et al. 2011). Furthermore, seabirds play a special cultural role on islands around the world, providing people with food (Circumpolar Seabird Working Group 2001), income (Duffy 1994), and cultural cohesion and identity (Lyver et al. 2008). Because most seabirds have nested for millennia on islands free of mammalian predators, they lack behavioural and life-history adaptations to avoid ground-based predation (Milberg and Tyrberg 1993). Thus, seabirds are vulnerable to alien predators, leading to species extirpation or severe population reduction on invaded islands (Towns et al. 2011). Seabird population declines have transformed entire island socio-ecological systems and their recovery is often integral to successful restoration (Croll et al. 2005; Fukami et al. 2006; Young 2014).

Predator eradication has become a prevalent seabird island restoration technique worldwide, with over 900 islands cleared of predators as of 2011 (Keitt et al. 2011; DIISE 2015). Following eradication of alien predators, restoration efforts can follow two basic approaches: passive, where no further intervention occurs and seabird populations and islands are assumed to recover naturally, and; active, where recovery of seabirds is enhanced by manipulating distribution and demography (Hobbs et al. 2011). Passive management operates under the "Field of Dreams" hypothesis, assuming that removing a stressor is sufficient to restore habitat and thus the capacity of seabirds to recover ("if you build it, they will come"; Palmer et al. 1997). It is generally cheaper than active management, but relies exclusively on the unpredictable capacity of populations and ecosystems to recover on their own (Scott et al. 2001; Jones and Schmitz 2009). Passive seabird population recovery is assumed to be slow because of k-selected life history characteristics, seemingly high philopatry, and intermittent breeding (Kappes and Jones 2014). Active management can help overcome impediments to population recovery, but often has high logistical and financial demands and variable success rates (Holl and Aide 2011; Jones and Kress 2012). Despite their logical interdependency, passive and active seabird island restoration techniques have evolved largely independently, so that few guidelines are available to help managers decide when to support passive recovery with active techniques.

Like most restoration endeavors, seabird island restoration is complex, operating within a dynamic network of ecological, cultural, social, and economic ideologies. Thus, deciding whether to employ active techniques would benefit from using a structured decision-making framework (Noss et al. 2009). Structured decision-making frameworks organize information, define objectives, and identify outcome alternatives and uncertainty, allowing restoration options to be weighted objectively (Wyant et al. 1995). A typical decision-making process involves: (1) outlining an explicit restoration goal; (2) building a conceptual model of the system; (3) identifying the most effective management approach; and (4) implementing and monitoring outcomes (Possingham et al. 2001). Seabird island restoration is unique in that there is an abundance of pre-existing passive and active restoration projects (Jones and Kress 2012; DIISE 2015). This represents an extraordinary opportunity to incorporate large-scale retrospective analyses of restoration outcomes directly into the decision-making framework. In this way, the amount of information about the recovery process is maximized, allowing for a more complete conceptual model of the system, and specific decision criteria to guide the prioritization of passive versus active management approaches before a project is implemented.

The objective of this study was to summarize how retrospective analyses of existing restoration data enhance a typical decision-making process, in particular for guiding when to invest additional time and money into active management. To illustrate this approach in a seabird island restoration context, we use the New Zealand archipelago as a case study. We focus on New Zealand due to the prevalence of burrowing seabirds, seabird-driven island systems, the country's rich history of predator eradications and island restoration, and the cultural significance of seabirds to Māori, New Zealand's indigenous peoples (Moller et al. 2000; Taylor 2000; Mulder et al. 2011a; Towns 2011; Buxton et al. 2014).

General decision-making framework

Decision-making frameworks are not new to prioritizing restoration interventions, but to our knowledge, are rarely used in island restoration (Helmstedt et al. 2016). Moreover, the formal use of previous restoration outcomes in decision frameworks when evaluating active versus passive management options is rare, notably for animal communities (but see Richardson et al. 2009 for decision-making framework for assisted colonization). We outline how initial analyses of existing restoration project outcomes can make each step of the decision-making process more efficient:

(1) Outline SMART project goals/objectives: Setting explicit goals and objectives is the integral starting point of restoration planning. By defining “SMART” (specific, measurable, attainable, relevant, and time-bound) goals and assessing the likelihood of achieving them at the outset of a restoration project, planning becomes efficient and step-wise, and allows measurement of success and return on investment (Hobbs et al. 2011). Assessing the effectiveness of previous restoration alternatives can help refine more robust and widely-accepted current restoration objectives. For example, by basing current objectives on past restoration efforts with high success rates, restoration plans are more likely to be cost-effective and thus receive financial and public support (Tunstall et al. 1999; Wilson and Bruskotter 2009; Kettenring and Adams 2011);

(2) Generate a conceptual systems model: In a restoration context, conceptual models outline a set of factors influencing the restoration target (i.e. population, community, or ecosystem), providing a simple visual or mathematical means to examine how a system may respond to management interventions (Heemskerk et al. 2003; Margoluis et al. 2009). Studies of chronosequences or comparative analyses of previous restoration outcomes can be used to gain insight into the mechanics of the recovery process, namely the natural ability of species and ecosystems to recover after passive restoration management;

(3) Project design– identify and prioritize management approaches: Restoration management interventions range from removal of the primary stressor to construction of novel ecosystems (Hobbs and Cramer 2008). Passive management may consist of little more than monitoring, as defined by the life histories of the system components being monitored. Active management is becoming more common at highly disturbed sites where the approach is feasible. Where more than one approach is available, the choice of which management intervention should be used on each species, site, or ecosystem requires a structured analysis of the probability of success of each alternative approach (Tear et al. 2005). Attributes affecting the probability of natural recovery can be identified and tested empirically to determine their relative contribution to a restoration approach successfully achieving the desired outcome (LoSchiavo et al. 2013). In many cases, collecting field data to model how a system may respond to a management intervention is not possible, either because the response to restoration is beyond the timescale of a current project or there are few comparable reference sites. Thus, a retrospective study of the successes and failures of similar projects, if available, can be a valuable source of information from which a set of key management decision criteria can be identified;

(4) Implement project and monitor outcomes: Monitoring outcomes of a restoration approach allows informed revision of project goals, conceptual models, and prioritization of future activities (i.e. adaptive management; Westgate et al. 2013). A project which is established as adaptive will be subject to fewer confounding factors than one that uses retrospective analyses of restoration outcomes (Armstrong et al. 2007; Lindenmayer et al. 2011). However, given the time frames associated with ecosystem recovery, adaptive management is likely to be less demonstrably effective in the short-term, highlighting the value of examining data from previous restoration projects (Lewis 2000; Jones and Schmitz 2009).

Applying the framework: New Zealand seabird island restoration

We illustrate a decision-making framework that incorporates retrospective analyses using the restoration of seabird-dominated island ecosystems in New Zealand. With an abundance of pre-existing restoration projects and a clear connection between seabird populations and island ecosystem functioning, a comparative analysis of seabird population response to eradication around the New Zealand archipelago can provide information to guide subsequent seabird island restoration management decisions.

New Zealand has over 700 islands larger than 1 ha and no native land mammals other than two extant bat species (Parkes and Murphy 2003). Since human colonization, over three-quarters of these islands have been modified substantially through burning, clearing, and the introduction of alien vertebrate species (Parkes and Murphy 2003; Bellingham et al. 2010). Seabirds were once abundant and widespread around New Zealand, which still has the highest diversity of seabird species in the world (Taylor 2000). However, predator introduction has resulted in the extinction, extirpation, or severe reduction of seabird populations (Holdaway and Worthy 1994; Taylor 2000; Veitch et al. 2004). Marine nutrient subsidies and in particular soil bioturbation by seabirds are central to island ecosystem functioning, and the loss of seabird populations due to predation has transformed ecosystem structure (Fig. 1; Mulder et al. 2011a; Orwin et al. 2015).

New Zealand is acknowledged internationally as a leader in island conservation and pest eradication with over 100 islands around the archipelago cleared of all alien animals (Towns et al. 2013). The country also has a long history of active restoration, including nearly 260 species transfers since the 1960 s, 16 of which were seabird species (Craig et al. 2000; Miskelly and Powlesland 2013).

Because of the widely recognized damage caused by alien mammals in New Zealand, the public is highly aware of conservation issues and community groups are increasingly involved in eradication and restoration (Forgie et al. 2001). However, general understanding of the role and diversity of seabirds is low (Seabrook-Davidson and Brunton 2014). Moreover, increasing co-governance of natural resources by Māori in New Zealand has led to greater consideration of their values, customary approaches, and practices within island management systems (Newman and Moller 2005; Lyver et al. 2009; Lyver et al. 2015a). Some Māori tribes harvest the chicks of sooty shearwaters (*Puffinus griseus*) or grey-faced petrels (*Pterodroma gouldi*), which are an important source of food, trade, cultural identity, and social cohesion (Taiepa et al. 1997; Lyver et al. 2008; Moller 2009). Historical relationships with the Pacific rat or kiore (*Rattus exulans*) add further complexity to island restoration planning. Kiore were brought to New Zealand by Polynesians c. 1280 (Wilmshurst et al. 2008), used by Māori as a food source, and feature prominently in their traditions, proverbs, and prayer (Haami 1993). Evidence suggests that kiore suppress native flora and fauna and that their eradication results in recovery of threatened species (e.g. Towns 2009). Yet for some Māori, the extinction of kiore would represent a the loss of a cultural treasure (Chanwai and Richardson 1998).

New Zealand's Department of Conservation (DOC) is charged with managing the country's biological and historic heritage. DOC's mandates include active management interventions, including restoration of island ecosystems; advocacy; and integrating the rights of local Māori (Towns et al. 1990b). However, DOC has undergone repeated large-scale restructures ever since its inception. The most recent, and most profound, was in 2011–2013, which included a structural model aimed at increased participation from business, non-government agencies, and community groups (Bushnell and Pratt 2014). As

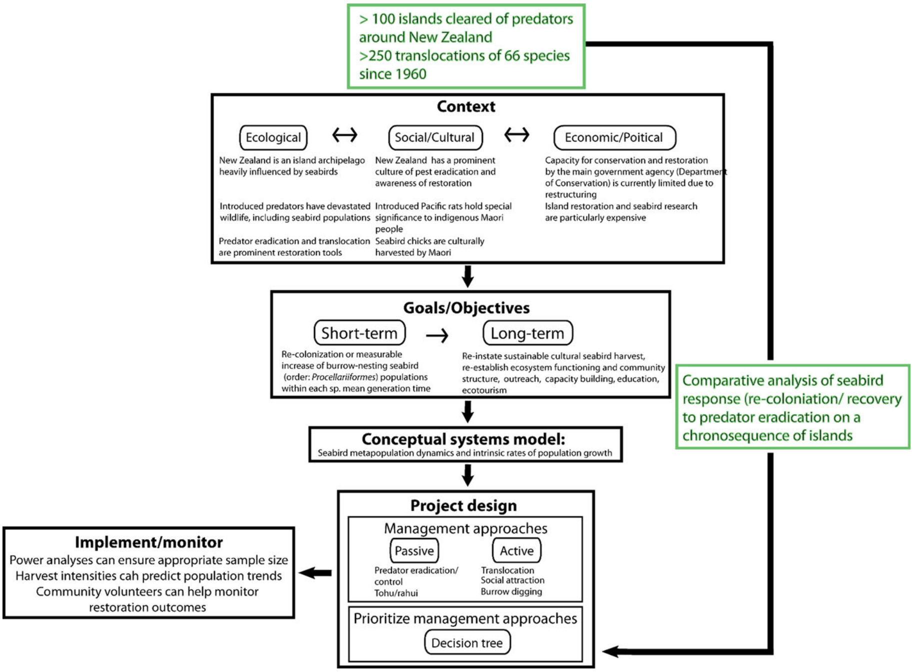

Fig. 1 Seabird island restoration in New Zealand to illustrate use of decision framework. The ecological, social and cultural contexts of the proposed project are summarized at the outset, generating a long-term SMART goal. A conceptual model of burrow-nesting seabird population recovery, constructed using models of metapopulation dynamics and information on drivers of intrinsic population growth, guides key decision criteria. To choose management approaches, these criteria predict whether natural recovery of burrow-nesting seabirds is likely and, if not, which active management methods should be used. Alternative approaches can then be compared for likely cost-effectiveness using a simple prioritisation metric. Both short- and long-term outcomes should be monitored, using a robust sampling scheme to report progress and to manage the project adaptively over time. (TEK = traditional ecological knowledge)

a result, DOC may delegate restoration projects to outside agencies more frequently than in the past. Given the high cost of seabird island projects, this could have consequences for the future financial security of island restoration.

Restoration goals

As the number of New Zealand islands from which alien predators have been eradicated has increased, island restoration goal setting has evolved. To illustrate this shift, we collated restoration goals from islands with predators eradicated around New Zealand from 1934 to 2011 (Table 1; Supplementary material S1). Using an updated list of predator eradication projects (Buxton et al. 2014), we searched for goals in published literature, restoration plans, threatened species plans, and eradication plans. The most common early (prior to 2000) goals of eradication were to improve eradication techniques, protect key species, or enable recovery of native species by removing predation pressure. Within the past decade, goals such as working with Māori to promote and support cultural values and aspirations (i.e. ‘bi-cultural management’ and ‘protect and conserve cultural aspects’; Table 1), education and outreach, restoring ecosystem functioning, and creating self-

Table 1 Summary of reported post-predator eradication goals of island restoration initiatives in New Zealand in published literature and technical plans between 1936 and 2011. Among all years and islands where the goal was reported, the 'mean report date' represents the average

Context	Goal	Total reports	Earliest report	Most recent report	Mean report date
Ecological	None	10			
	Test eradication techniques	37	1978	2012	1994
	Protect specific species	51	1970	2008	1995
	Determine the impacts of predation	8	1979	2005	1996
	Seabird recovery/restoration	21	1985	2014	1996
	Reintroduce natural flora and fauna	44	1970	2014	1996
	Restore communities	11	1990	2012	1997
	Reintroduce functional groups	1	1999	1999	1999
	Enable recovery by removing predation pressure	63	1946	2014	2000
	Research reintroductions	3	1996	2007	2000
	Restore pre-human state	3	1999	2003	2000
	Create a refuge for threatened species	63	1970		2000
	Reforestation	14	1982	2014	2003
	Create self-sustaining population of rare species	4	1999	2012	2007
Economic	Ecotourism	6	1982	2012	2001
	Restore ecosystem functioning	21	1990	2014	2003
Social	Community participation	10	1990	2014	2001
	Education outreach	29	1982	2013	2004
	Bi-cultural management	22	1978	2013	2005
	Reinstate sustainable cultural harvest of seabirds	2	1998	2012	2005
	Protect and conserve historic sites	22	1990	2014	2006
	Protect and conserve cultural aspects	11	1978	2012	2009

sustaining populations of rare plants and animals have become more prevalent. Generally, we found few goals that conformed to SMART criteria.

In the earlier days of ecological restoration planning in New Zealand, goals were dominated by a Eurocentric historical perspective, where the endpoint of a restoration program was a biotic community that was assumed to represent the pre-human past (Atkinson and Towns 1990; Towns et al. 1990b). However, palaeoecological investigations indicate that, before human colonization, some offshore islands were dominated by podocarp forests which have no modern analogue (Wilmshurst et al. 2014). This evidence, and the cultural significance of islands for Māori, suggests that pre-human island restoration targets may be neither feasible nor desirable (Bellingham et al. 2010; Lyver et al. 2015b). Thus, contemporary post-eradication island restoration goals in New Zealand include: maintaining ecosystem processes, preventing extinction, improving species and functional diversity, forming partnerships with local Māori, preserving historic and cultural values, and fostering community engagement, enjoyment, and recreational use (Table 1; Lee et al. 2005; Department of Conservation 2010).

Although seabird population recovery is likely to be a significant component of attaining these modern island restoration goals (Moller 2010; Mulder et al. 2011a), few early restoration initiatives acknowledged its importance (Table 1). Appropriate baseline data were therefore rarely collected to allow seabird recovery outcomes to be assessed. Because seabirds, especially petrels and shearwaters, exert a large influence on island ecosystems in New Zealand, we propose that seabird re-colonization and colony growth may be beneficial SMART post-eradication outcomes for many offshore islands (e.g. see Imber et al. 2003). A project may select a particular species, the recovery of which is measurable within the respective species' generation time (approximately 15 years for petrels and shearwaters; IUCN 2012). Longer-term (e.g. >30 years) outcomes for seabird-driven island restoration could include re-establishing island ecosystem functioning and community structure, and, on some islands, the re-instatement of cultural harvest (Fig. 1).

Conceptual systems model for the restoration of NZ's seabird islands

Assuming that the restoration goal of seabird recovery is selected, reliable prediction of the response of seabird populations to management interventions at breeding sites requires a basic understanding of local seabird population dynamics (Margoluis et al. 2009). In this way, a conceptual model of seabird recovery can be constructed by identifying the drivers of population growth (Buxton et al. 2014).

Generally, seabirds form metapopulations, with breeding sites separated by water barriers and where local population growth rates are affected by both intrinsic and extrinsic dynamics (McCullough 1996; Matthiopoulos et al. 2005). Intrinsically, seabirds have low annual reproductive output, fecundity is low, and intermittent breeding is common, resulting in low rates of per capita growth (Warham 1990; Cubaynes et al. 2011). Intrinsic negative density-dependence may result from limitations in breeding sites and food (Croxall and Rothery 1991; Sandvik et al. 2012), while positive density dependence may be associated with coloniality (Kildaw et al. 2005). Moreover, despite being highly mobile, behavioral mechanisms associated with coloniality (e.g. philopatry and social attraction) mean that the number of immigrants recruiting from other colonies is thought to be low (Hamer et al. 2002). Thus, population dynamics are likely to be slow and characterized by traits that may not conform to metapopulation theory (Matthiopoulos et al. 2005). When predators are removed from an island, local seabird population growth (i.e. passive recovery) will depend on a number of factors; for example, the size of the remnant population at the time of eradication, species-specific life history traits (such as age of first reproduction), local habitat quality, and density-mediated immigration rates. However, because of seabirds' slow population growth, evaluating how these parameters drive population recovery is problematic within the time constraints of a restoration project.

Towns (2002) and Towns et al. (2012) recommend long term environmental monitoring of islands that escaped predator invasion to build conceptual models of recovery. In the absence of robust long-term monitoring data, chronosquence analyses or 'space-for-time substitutions' have been used to compare ecosystem function or population dynamics among islands with different periods since pest eradication (Fukami et al. 2006; Jones 2010a; Jones 2010b). Although caution must be taken when examining successional processes using chronosequences (Johnson and Miyanishi 2008), if results can be validated from data using other methods (e.g. monitoring before and after eradication), they could represent a useful proxy for temporal recovery dynamics (Towns 2009). Given the abundance of eradication projects around New Zealand, chronosequences have been useful in identifying factors driving post-eradication seabird response (Buxton et al. 2014).

Generally, we have a poor understanding of how seabirds and ecosystems respond to introduced predator eradication and active restoration (Mulder et al. 2009). In the past, threatened species conservation has assumed greater urgency than consideration of the impact of such species on the island's existing biota. For example, the reintroduction of natural seabird predators (e.g. tuatara *Sphenodon sp.*) at too early a stage in recovery may result in reduced probability of the re-establishment of small seabird species (Atkinson and Towns 1990; Towns et al. 1990a). Moreover, if seabird predators are introduced before the recovery of their seabird prey, this may result in lower probability of establishing the predator itself.

Management techniques

The most widespread and effective passive seabird island restoration technique in New Zealand is the eradication of alien predators (Towns and Broome 2003; Clout and Russell 2006; Broome 2009). In some cases, abundant native avian predators may also prey on breeding seabirds, and populations must be controlled to reduce their impact on threatened seabirds (Miskelly 2013). The temporary cessation, *rāhui,* of traditional harvest from seabird islands where Māori are owners or have customary rights may also be employed to assist population maintenance or recovery (Moller 2006; Kitson and Moller 2008; Lyver et al. 2015a).

After eradication, a number of active techniques can be used to restore habitat and encourage seabird population recovery. Both exotic weed control and revegetation of indigenous plant communities can facilitate seabird recovery actively through habitat enhancement (Towns et al. 1997; Forbes and Craig 2013). Chick translocation to a restoration site, before the age where they imprint to their natal site, has been successful for a number of seabird species (Jones and Kress 2012). However, because of the high cost and labor requirements of transportation and feeding, plus an average lag time of five years until birds return to a restoration site to breed, translocation is expensive and outcomes are difficult to predict (Miskelly et al. 2009). Social attraction (e.g. playback of recorded vocalizations or decoys; Kress 1998), where birds are lured to formerly-occupied or new breeding habitat by mimicking social cues is emerging as a cost-effective alternative technique on some islands in New Zealand (Sawyer and Fogle 2010). Because remnant colonies of seabirds may have skewed species composition, resulting inter-specific competition may require active management (Buxton 2014). For example, widespread interference competition between broad-billed prions (*Pachyptila vittata*) and endangered Chatham petrels (*Pterodroma axillaris*) on South-east Island, New Zealand, poses the most serious threat to the latter (Was et al. 2000). Restoration managers have intervened by culling broad-billed prions and constructing Chatham petrel nest-boxes that exclude broad-billed prions (Gummer et al. 2015). Similarly, an active technique used by some Māori tribes has involved the splitting of existing burrows or drilling of new ones to increase breeding habitat for burrowing petrels (Lyver et al. 2008).

Prioritizing management approaches

Assuming the goal of restoration is to encourage recovery of seabird populations, a series of decisions will need to be made relating to how population recovery can best be achieved. Managers will first need to consider a conceptual model of the key factors influencing natural seabird population recovery. This includes factors influencing the recruitment of individuals from other islands to the newly predator-free space as well as the intrinsic rate of growth of any remnant colony. We used a previously published comparative analysis of over 100 seabird population responses to predator eradication on 41 islands (Buxton et al. 2014) to identify key decision criteria in choosing between active and passive management alternatives (Fig. 2). These criteria are:

Distance to source population

Colony growth and re-colonization after eradication is reliant on the size, distance, and connectivity to nearby source populations (Hanski 1998). Evidence suggests that recruitment probability decreases exponentially with distance from a source population, and that re-colonization is unlikely without a colony within ~25 km (Oro and Pradel 1999; Buxton et al. 2014).

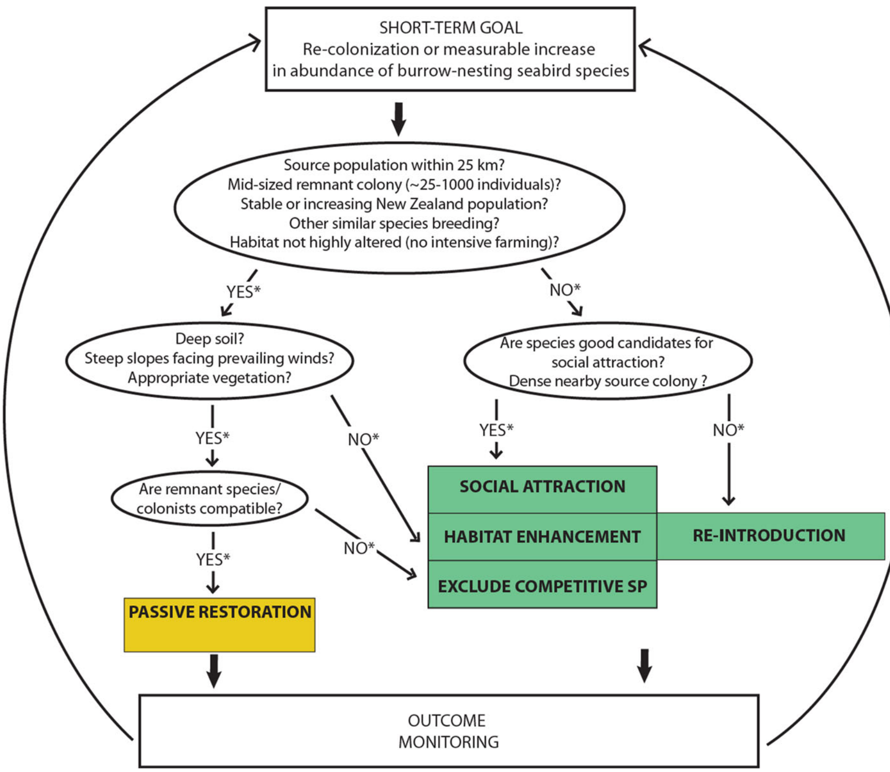

Fig. 2 Ecological decision tree to guide management interventions for burrow-nesting seabirds based on the probability of natural recovery following predator eradication. Active (*green*) versus passive management (*yellow*) can be chosen based on site- or species-specific characteristics. *Asterisks* indicate that the majority of criteria are met ("yes") or not met ("no"). Further considerations include cost and stochastic events. (Color figure online)

Remnant colony size

seabird colony growth is regulated by both positive density dependence, where birds are attracted to settle in larger pre-existing colonies, and negative density dependence, where crowding decreases habitat quality at higher densities (Kildaw et al. 2005; Buxton et al. 2016b). Thus recovery is more likely in a mid-sized remnant colony ($\sim$25–100 breeding pairs; Buxton et al. 2014).

Metapopulation status

Growth of a local population after eradication is more likely if the species' overall metapopulation is stable or increasing in size (Oro 2003).

Presence of other breeding seabird species

Not only are seabirds more likely to settle amongst colonies of conspecifics, colonies of other species may also be attractive; re-colonization is more likely on islands with higher seabird species richness (Parejo et al. 2005; Buxton 2014). This relationship may reflect habitat quality, where recovery is more likely on islands with less habitat alteration (e.g. no grazing by cattle), or heterospecific habitat copying, where prospecting individuals of one species respond to the presence of other species with similar ecological needs (Wagner et al. 2000).

If a potential restoration site does not meet these criteria for the focal species, active methods should be considered. Some species and sites may be better candidates for a less logistically demanding active approach (e.g. social attraction; Fig. 2). For example, grey-faced petrels readily disperse from their natal site and will settle and breed at sites with call-playback (Sawyer and Fogle 2010; Lawrence et al. 2014; Buxton et al. 2015b). For grey-faced petrels and fluttering shearwaters (*Puffinus gavia*), responses to playback increases if larger, denser colonies are closer to the restoration site. In instances where species exhibit strong site fidelity, the restoration site is far removed from a source population, or species are critically threatened and require immediate intervention to prevent extinction, chick translocation is more likely to achieve recovery objectives (Fig. 2; Miskelly et al. 2009). Conversely, the biology of some species may inhibit the success of active restoration (e.g. unfeasibly complex post-fledgling care or diet) and must also be considered during prioritization.

The ability of a remnant colony to grow will also depend on local habitat suitability. Although burrow-nesting seabirds' nesting habitat selectivity can weaken as a colony grows, birds use deeper soils, steeper slopes facing prevailing winds, and later-succession vegetation types more readily than other habitat characteristics (Whitehead et al. 2014; Buxton et al. 2015a). Some features of a proposed restoration site might preclude consideration of restoration or demand intensive preparation (e.g. revegetation) prior to any attempts at species manipulation. Once a species is established, the nature and intensity of interactions between sympatric species in a community can vary, particularly if selective predation pressure has skewed community structure (Buxton 2014). Some burrow-nesting seabird species may competitively exclude others, especially when communities are re-assembling after eradication (e.g. grey-faced petrel and little shearwater Puffinus assimilis Pierce 2002; Buxton 2014). In this case, some species may remain rare because they cannot withstand inter-specific competition, especially if the population of the dominant species continues to increase (Oro et al. 2009). These factors must be considered in advance of any decision to embark on a project (Fig. 2).

At this point in the planning process, the probability of restoration success should be analyzed in a bioeconomic framework to weigh cost and benefit (see Joseph et al. 2009; Jones and McNamara 2014 for details of accessible methods).

Implementation and monitoring

Although monitoring is an integral component of management activity, the outcomes of eradication are rarely measured in New Zealand (Jones et al. 2011). If SMART project outcomes are defined clearly, indicators of success at each stage should be easily identifiable and monitoring can be included as part of the project planning. Without appropriate monitoring it is impossible to determine whether or not the project has been a success or to manage the project adaptively through time. Moreover, monitoring data can help validate predictions made by analyzing recovery across chronosequences.

For seabirds, there are numerous logistical challenges that preclude simple monitoring strategies, including cryptic nesting behavior (below-ground nesting and nocturnal colony attendance; Warham 1990), high costs of getting to island breeding grounds, and high interannual variability in breeding participation (Newman et al. 2009). However, new technologies are being applied to seabird monitoring, including infra-red burrow camera surveys (Hamilton 2000), automated acoustic sensors (Buxton and Jones 2012; Borker et al. 2014), and radar (Gauthreaux and Belser 2003), while modern simulation techniques allow greater refinement of monitoring data (Buxton et al. 2016a). Moreover, there is a well-established tradition of community involvement in conservation in New Zealand (Peters et al. 2015) and an increasingly strong recognition of the role of Māori traditional knowledge in monitoring seabird populations (Kitson 2004; Moller et al. 2004). Participatory approaches are also valuable, as public involvement enhances restoration acceptance, increases the capacity for active restoration through volunteering, and builds capacity through education and outreach (Parkes and Panetta 2009; Hardie-Boys 2010).

Discussion

Deciding which restoration techniques to implement on islands is complicated: restoration goals are steeped in conflicting socio-political and cultural contexts, recovery trajectories of species are difficult to predict, restoration is expensive, and budgets are limited. Decision-making tools are well-suited to help managers decide objectively between restoration alternatives under complex circumstances. When it comes to decision-making, island systems are at a unique advantage– a wealth of previous restoration outcome information exists that can be incorporated readily into decision-making frameworks. Accordingly, the formal inclusion of retrospective analyses of species response to past restoration approaches can contribute to several stages of a decision-making framework: successes can increase the socio-economic acceptance of a project, while comparative analyses can identify factors driving species recovery and determine the likelihood of the success of alternative restoration approaches. Identifying a set of key criteria based on previous restoration data can help create step-wise decision process to select among restoration approaches, without the need for further field data to be collected.

Due to the prevalence and success of predator eradication around New Zealand, a comparative analysis of seabird population responses after eradication revealed drivers of recovery, site characteristics, and species where natural recovery is more likely. Conversely, this indicated situations where active intervention should be prioritized. Retrospective analyses require results from previous projects to be made available, even if those projects were unsuccessful. Accordingly, this approach depended on the 'grey' literature, given the bias in publishing only success stories (Csada et al. 1996). Decision-making within the seabird island restoration process would be based, ideally, on assessment of passive and active restoration outcomes planned initially as tests of alternative approaches (i.e. adaptive management; Williams 2011). However, the timeframe required for this adaptive approach may be unrealistic for restoration decisions involving long-lived seabirds. The existence of numerous pre-existing passive and active restoration projects means that their outcomes can be incorporated into prioritizing current restoration approaches, while monitoring outcomes of the current project can validate or refine predictions made using previous projects. The seabird island decision framework outlined in this paper can guide managers' choice of tools to facilitate seabird recovery.

Since its inception as a discipline in conservation biology 60 years ago, island restoration has evolved from a field that was dominated by removing threats and assuming that populations will eventually recover, to one where the recovery of ecosystems and wildlife can be facilitated actively and outcomes monitored. As we move forward with restoring complex island communities, decision-making tools using a logical, step-wise framework informed by previous restoration successes and failures will aid in increasing understanding and reducing uncertainty of ecosystem response.

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