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RESEARCH ARTICLE

A GIS-based decision-making approach for prioritizing seabird management following predator eradication

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Given that 29% of seabird species are threatened with extinction, protecting seabird colonies on offshore islands is a global conservation priority. Seabirds are vulnerable to non-native predator invasions, which reduce or eliminate colonies. Accordingly, conservation efforts have focused on predator eradication. However, affected populations are often left to passively recover following eradications. Although seabirds are highly mobile, their life history traits such as philopatry can limit passive recolonization of newly predator-free habitat. In such cases, seabird colonies can potentially be re-instated with active restoration via chick translocations or social attraction methods, which can be risky and expensive. We used biogeographic and species-specific behavioral data in the Hauraki Gulf, New Zealand, a global hotspot of seabird diversity and predator eradications, to illustrate the use of geographic information systems multi-criteria decision analysis to prioritize islands for active seabird restoration. We identified nine islands with low observed passive recovery of seabirds posteradication over a 50-year timeframe, and classified these as sites where active seabird management could be prioritized. Such spatially explicit tools are flexible, allowing for managers to choose case-specific criteria such as time, funding, and goals constrained for their conservation needs. Furthermore, this flexibility can also be applied to threatened species management by customizing the decision criteria for individual species' capacity to passively recolonize islands. On islands with complex restoration challenges, decision tools that help island restoration practitioners decide whether active seabird management should be paired with eradication can optimize restoration outcomes and ecosystem recovery.

Key words: active restoration, decision tools, GIS-MCDA, island conservation, prioritization, threatened species

Implications for Practice

- Spatially explicit GIS-MCDA can facilitate prioritization of active seabird management actions paired with predator eradication to better allocate limited resources.
- On seabird islands where there are multiple or complex restoration requirements, GIS-MCDA can help managers prioritize decisions using objective ecological information.
- GIS-MCDA is an accessible decision tool for island restoration or threatened species practitioners, which can be customized to integrate social, economic, and ecological factors of island restorations.

Introduction

Throughout the world, islands provide breeding habitat for seabirds, which generate distinctive ecosystems. Seabirds are allogenic ecosystem engineers because of the physical disturbance from nesting activity and the significant marine nutrient subsidies provided by guano, prey remains, corpses, and eggshell deposition (Jones et al. 1994; Mulder et al. 2011; Smith et al. 2011; Caut et al. 2012). However, seabirds are among the most threatened marine vertebrates; globally, 29% of species are at risk of extinction (Croxall et al. 2012; Birdlife International 2014). Life history traits such as coloniality, low reproductive output, extended periods of parental care, and slow growth to maturity make seabirds vulnerable to introduced predators (Jones et al. 2008; Towns et al. 2011; Croxall et al. 2012; Spatz et al. 2014). Over 90% of the world's islands have been invaded by rats (*Rattus* spp.) and at least 65 major island groups have been invaded by cats (*Felis catus*), making introduced predators one of the most large-scale and acute threats to seabird colonies (Jones et al. 2008; Croxall et al. 2012; Spatz et al. 2014). The loss or reduction of seabird populations from an island alters the allochthonous inputs of nutrients, affecting every trophic level of the ecosystem (Fukami et al. 2006; Jones et al. 2011). Accordingly, conservation efforts have focussed on predator removal from terrestrial breeding sites, henceforth

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called “seabird islands”, to recover seabirds and island biodiversity alike (Howald et al. 2007; Towns et al. 2009; Bellingham et al. 2010; Keitt et al. 2011).

Following predator eradications, extirpated seabird populations can either recolonize naturally (passive recovery) or through active restoration. The latter involves facilitated colonization through methods such as chick translocations, acoustic playback, decoys, and mirrors (Jones et al. 2011; Jones & Kress 2012). Numerous challenges are associated with active seabird management actions including high failure rates, substantial resource needs, and complex logistical demands (Ricciardi & Simberloff 2009; Suding 2011). Therefore, passive recovery of seabird populations is the most common option following predator eradication (Mulder et al. 2009; Suding 2011).

Some seabird species may fail to recover or establish new breeding sites following predator eradication, due to life history traits such as philopatry and conspecific attraction (Warham 1990; Schreiber & Burger 2001; Anderson & Mulder 2011; Kappes & Jones 2014). The combination of rare emigration and low reproductive output means that for many seabird species, the recovery of posteradication colony may take decades or longer, if it happens at all (Buxton et al. 2014; Kappes & Jones 2014). For example, cats were removed from Cuvier Island 53 years ago and Pacific rats (*Rattus exulans*) were eradicated 22 years ago (Towns & Broome 2003; Towns et al. 2013). However, the recovery of remnant seabird colonies has been slow and there has been no recolonization by seabirds from neighboring islands (D.R. Towns unpublished data).

The World Conservation Union (IUCN) best practice guidelines for restorations emphasize that restoration actions should focus on the re-instatement of species that play a disproportionate role in ecosystem function (IUCN 2012). Evidence suggests that, because of their ecosystem engineering role, recolonization of extirpated seabird species or enhancing population growth of remnant colonies is necessary to restore island ecosystem function following predator eradication (Croll et al. 2005; Buxton et al. 2014; Kappes & Jones 2014). Island restoration practitioners may assume that by eradicating predators and making habitat available, seabirds will passively recolonize, thereby reactivating lost ecosystem drivers (Kappes & Jones 2014). Although a return to a seabird-driven system from passive recovery has occurred in some cases (e.g. Korapuki Island; Towns 2002), it has rarely been the outcome (Jones et al. 2011; Kappes & Jones 2014). The incomplete recovery of seabird colonies will fail to re-instate drivers of seabird island ecological processes, likely resulting in systems with an alternative stable state (Mulder et al. 2009; Jones 2010*a*; Kappes & Jones 2014).

The uncertainty and complexity of seabird behavior and population dynamics can lead to unpredictable outcomes following active seabird restoration actions (Towns 2002; Keenelyside et al. 2012). Chick translocations and acoustic attraction methods can have particularly high failure rates, if there is not a thorough understanding of life history traits of the target species (Jones & Kress 2012; Buxton et al. 2015*b*). Furthermore, the relationship between seabird life histories and island ecosystem function, confounded by poor data about pre-invasion island ecology and historical seabird populations and a lack of comparable reference sites, means that island restoration projects are sometimes likened to “reconstructing the ambiguous” (Simberloff 1990, p. 37; Towns 2002; Suding 2011). Furthermore, active seabird management becomes increasingly complex for threatened seabirds, where little is known about the ecology of the species. For example, the New Zealand storm petrel (*Fregetta maoriana*), once thought extinct was discovered breeding on Hauturu in 2013, and at present too little is known about the ecology of the species to inform management actions (Gaskin & Rayner 2013). Restoration decisions for any degraded system or species-specific management require careful evaluation to maximize resource allocation and improve the likelihood for success (Suding 2011; Jones & Kress 2012). Decision-making tools are needed that can allow island restoration and threatened species practitioners to weigh active versus passive options. In this way, the probability of achieving a restoration goal such as threatened species management or re-instatement of seabird island ecosystem function can be optimized, whereas cost is minimized (Kappes & Jones 2014).

We used biogeographic data from the Hauraki Gulf, New Zealand, a seabird diversity hotspot with a history of predator eradications to illustrate the application of geographic information systems multi-criteria decision analysis (GIS-MCDA). GIS-MCDA is an accessible, user-friendly tool that allows users to determine weights of multiple variables (e.g. habitat suitability, interspecific interactions, and value-based restoration goals) that are known to influence recovery. We used GIS-MCDA to illustrate how habitat associations, and behavioral and life history drivers of passive recolonization can be combined to better predict natural seabird recovery potential after eradication. We use the resulting probability of passive recolonization to make recommendations of where managers should prioritize active seabird restoration in the Hauraki Gulf and how this method can be used generally for spatial prioritization of active seabird restoration actions.

Methods

Study Sites and Species

The Hauraki Gulf, New Zealand, supports breeding populations of 27 seabird species (Gaskin & Rayner 2013). Here, we focus on the most abundant and diverse order of seabirds in the region, Procellariiformes, which comprise burrow-nesting petrels and shearwaters (14 species, Table S1 of Appendix S1, Supporting Information). These species exhibit colonial breeding characteristics, and have a disproportionate influence on island habitats in New Zealand through allochthonous nutrient inputs and burrowing activity (Jones 2010*a*; Gaskin & Rayner 2013; New Zealand Birds Online 2013). The climate of the Hauraki Gulf is temperate-humid with a mean annual temperature of 16°C, relative humidity of 87%, and a mean average rainfall of 1,202 mm (CliFlo 2014). We use seabird presence and population estimates from 229 records of 14 Procellariiformes on 69 island sites (C. Gaskin unpublished data, Appendix S2), which included islands cleared of introduced predators spanning

78 years (1936 to 2014; $n = 31$; Table S2 of Appendix S1) and islands that have never been exposed to introduced predators ($n = 37$; Table S3 of Appendix S1). Predator presence data for offshore islands in the Hauraki Gulf were collated from Taylor (1989) and D.R. Towns (unpublished data).

Quantitative Analysis. All statistical analyses were done in R statistical software v. 3.1 (R Core Team 2013). To determine the expected number of seabird species as a function of island area, we used quantile regression analysis (95th, $p < 0.05$) of species richness by island area with a subset of islands that had never been exposed to introduced predators (Table S2 of Appendix S1; *quantreq* package, Koenker 2009). We compared species richness of predator-free islands to islands with predators eradicated (islands target for restoration, hereafter "target islands"). If target islands had fewer than the expected seabird species richness of uninvaded islands, we included them in the MCDA (Table S3 of Appendix S1). Island area was log transformed for all statistical analyses.

Decision Criteria. Seabird recruitment to new breeding sites is influenced by metapopulation dynamics, habitat availability, and social cues (McGowan et al. 2013; Buxton et al. 2014). We defined high-priority sites for assisted seabird colonization as those that lacked remnant seabird colonies (Parejo et al. 2005), and were beyond the most probable natural recolonization distance of source populations (Buxton et al. 2014). We used ArcGIS version 10.1 (ESRI 2011) for GIS-MCDA to aggregate quantitative and qualitative data to explore seabird restoration decisions in a heuristic manner.

We used four main environmental and geographic criteria (listed below) and GIS-MCDA to identify islands with suitable habitat features but a low likelihood of passive seabird recolonization (Buxton et al. 2014). We took a risk-averse conjunctive approach, where all criteria in the analysis must be met to be included in the final output (Greene et al. 2011). We gave an even weighting to each main criterion (25%). Within each of these criteria were several attributes (e.g. vegetation type, slope, particle size, and induration). Attributes were standardized using the Fuzzy Overlay geoprocessing tool, meaning that the combination of multiple attributes is more important or larger than any of the inputs alone (i.e. the site had to have all of the attributes to be identified as a priority site for active seabird management; Appendix S1). We ranked the islands on the summed score of the MCDA criteria, where each criterion were given a score between 0 and 1, where 1 is the most suitable site and 0 is unsuitable (Table 1).

Habitat Suitability. We quantified the availability of suitable nesting habitat by calculating composite vegetation and soil habitat parameters relevant for all procellariid species included in our analysis (Tables S3, S4, S5, & S6 of Appendix S1). We identified species ecology and breeding habitat associations from the literature, NZ Birds Online (2013), and BirdLife International (2013) (Table S1, Supporting Information). We used vegetation classifications from the New Zealand Land Cover Database (LCDB) version 3.0 (available at https://lris.scinfo.org.nz/layer/401-lcdb-v33-land-cover-database-version-33/, accessed 27 Jun 2014), and soil classifications from the Land Environments of New Zealand (LENZ) database (available at https://koordinates.com/search/?q=LENZ, accessed 27 Jun 2014).

We weighted each of the four habitat attributes in the GIS-MCDA evenly (25%): (1) vegetation, (2) soil quality, (3) induration, and (4) slope (Table S9 of Appendix S1). To calculate the percentage weights for each attribute in the habitat suitability criteria, we calculated the number of seabird species with each habitat criterion and divided this by total available habitat (Tables S5 & S6 of Appendix S1). The resulting weights used for the vegetation attribute were forest 45%, shrubland 41%, and grasslands 14%. The remaining attributes relating to soil qualities among associated burrowing seabirds were weighted toward friable, low induration soils on 5°–26° slopes (Table 1; Bancroft et al. 2005; Whitehead et al. 2014; Buxton et al. 2015*a*; Table S1 of Appendix S1). For example, gray-faced petrels (*Pterodroma gouldi*) are known to burrow in forests and grasslands, in sandy, low non-indurated soils on slopes between 10° and 40° (Whitehead et al. 2014). We kept the suitable habitat parameters broad to ensure all species habitats would be included, thus detail on specific habitat associations for individual species may be lost. However, the habitat criteria reflect the types of vegetation, soils, induration, and slope that species in our analysis are more frequently found to nest in (Tables S1 & S4 of Appendix S1).

Heterospecific Social Attraction. Passive recolonization events have been shown to be greater on islands with more than two seabird species present, likely due to heterospecific attraction to breeding habitat (Danchin et al. 1998; Parejo et al. 2005; Buxton et al. 2014). If there were fewer than two seabird species known to breed on an island, it was weighted as 1, or most suitable in the analysis (Table 1). Because none of the islands had the expected number of species present posteradication (Fig. 1), we assigned decreasing weight values as the number of species increased (Table S9 of Appendix S1).

Source Population Size. For philopatric seabirds, the attraction to an established breeding site can inhibit the formation of new colonies; however, individuals may disperse to newly available habitat when the costs outweigh the benefits of settling in an established colony (Kildaw et al. 2005). Our target islands were chosen as high-priority sites for assisted colonization if potential source populations were likely insufficient to result in emigration to the target islands (Buxton et al. 2014). Potential source populations for passive recovery were identified as colonies with more than 25 breeding pairs and included 11 of the 14 species in our analysis (Table S6 of Appendix S1). We increased the sub-criteria weight values of the source colony according to population size: 25–100 pairs (25%), 100–1000 pairs (35%), more than 1,000 pairs (40%), and zero for colonies less than 25 pairs (Table S9 of Appendix S1).

Table 1. Islands included in the GIS-MCDA to prioritize active seabird management. Membership scores for each criterion are a value between 0 and 1, where 1 is the most suitable site and 0 is unsuitable given the input criteria. The total MCDA score is the sum of each criteria membership score. Colony size and proximity criteria were calculated together giving a total MCDA score out of 3. The islands in italics are those identified as high priority for active seabird restoration (a total score of more than 1.5). The islands listed in bold are discussed as examples.

Island	*Area (ha)*	*Years Since Eradication*	*Habitat MCDA Score*	*Heterospecific MCDA Score*	*Source Colony Size and Proximity*	*Total MCDA Score*
Hauturu (Little Barrier)	***2817***	***10***	***0.99***	***0.99***	***0.99***	***2.97***
Rakitu (Arid)	***312.33***	***0***	***0.99***	***0.99***	***0.99***	***2.97***
Motuhoropapa	*8.6*	*50*	*0.99*	*0.99*	*0.94*	*2.92*
Tiritiri Matangi	*192.23*	*21*	*0.3*	*0.94*	*0.99*	*2.23*
Rangitoto	*2325.69*	*5*	*0.99*	*1*	*0.22*	*2.21*
Otata	*15*	*12*	*0.99*	*0.99*	*0.04*	*2.02*
Motutapu	*1558.6*	*5*	*0.99*	*1*	*0.02*	*2.01*
Motuihe	*180.52*	*9*	*0.86*	*1*	*0.12*	*1.98*
Rotoroa	*89.24*	*21*	*0.86*	*0.99*	*0.07*	*1.92*
Atihau (Trig)	17.24	23	0.99	0.5	0	1.49
Coppermine	76.82	17	0.99	0.5	0	1.49
Cuvier (Repanga)	**169.33**	**21**	**0.99**	**0.5**	**0**	**1.49**
David Rocks	0.31	50	0.74	0.03	0.61	1.38
Fanal	76.8	18	0.99	0.09	0	1.08
Whatupuke	98.94	18	0.99	0.09	0	1.08
Middle	21.76	20	0.99	0.04	0	1.03
Ohinau	41.96	9	0.99	0.04	0	1.03
Red Mercury (Whakau)	213.33	22	0.99	0.03	0	1.02
Kawhitihu (Stanley)	94.85	23	0.99	0.003	0	0.993
Mauimua (Lady Alice)	150.56	21	0.99	0.003	0	0.993
Taranga (Hen)	488.59	3	0.99	0.003	0	0.993

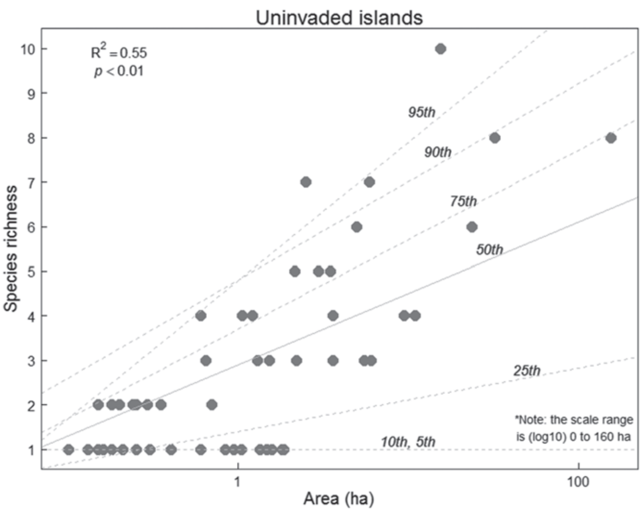


Figure 1. Seabird species richness–island area relations of islands that have remained free of introduced predators in the Hauraki Gulf Marine Park, New Zealand.

Proximity to Source Population. Evidence suggests that passive seabird recolonization is negatively related to distance to source populations. Beyond 25 km, the probability of passive seabird recolonization response falls below 50% (Buxton et al. 2014). Thus, we weighted islands within 25 km as zero and those with a source beyond 25 km as 1, meaning they were most suitable for active seabird management.

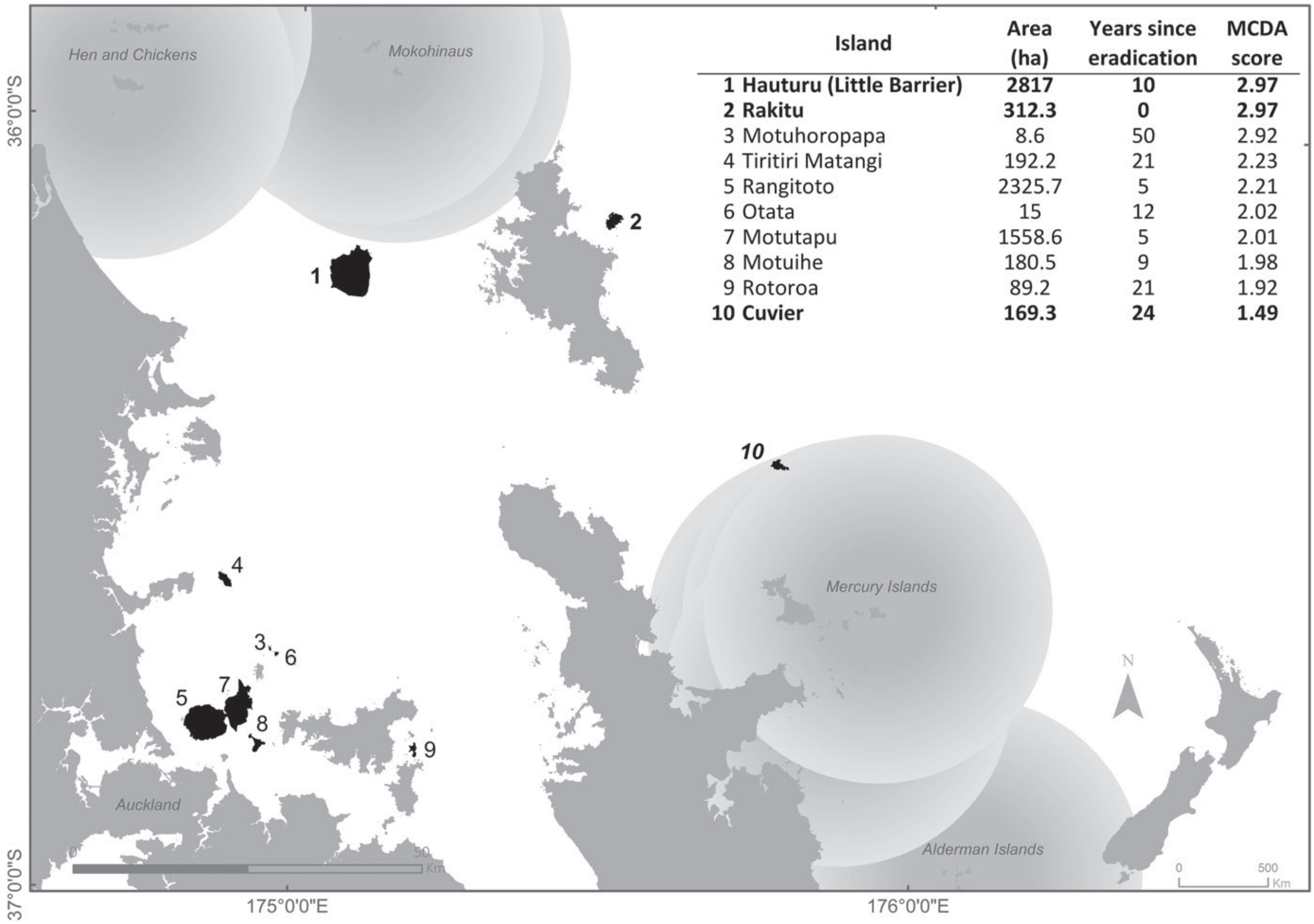
Results

We investigated the species richness–island area relationships of islands that have remained free of introduced predators in the Hauraki Gulf and found a positive linear relationship on a semi-log scale ($r^2 = 0.55$, $p < 0.01$; Fig. 1). Therefore, we excluded 10 islands recovering from predators, which met the expected number of species given island area relationship (Table S6 of Appendix S1), leaving 21 target islands for the GIS-MCDA (Table 1).

The time since the eradication of predators from the islands ranged from 5 years (Rangitoto and Motutapu: 2009) to 50 years (Motuhoropapa: 1964) and one island for which eradication is imminent (Rakitu: baiting began in 2014) (Table 1). Of the 21 target islands that did not have expected numbers of seabirds given island area (above), 17 met the GIS-MCDA habitat suitability criteria (membership values >0.86; Table 1). We identified 134 source populations on 42 islands (Table S8 of Appendix S1), and 9 islands that were not within 25 km of one or more of those source populations (Table 1). We designated these nine islands as high-priority sites for active seabird management (Fig. 2).

Discussion

Although introduced predator eradication is identified as an effective seabird conservation strategy (Towns et al. 2013), little research has investigated where and when actively restoring seabirds might be necessary once predators have been removed (Kappes & Jones 2014). If seabirds fail to recolonize or recover



Island	Area (ha)	Years since eradication	MCDA score
1 Hauturu (Little Barrier)	**2817**	**10**	**2.97**
2 Rakitu	**312.3**	**0**	**2.97**
3 Motuhoropapa	8.6	50	2.92
4 Tiritiri Matangi	192.2	21	2.23
5 Rangitoto	2325.7	5	2.21
6 Otata	15	12	2.02
7 Motutapu	1558.6	5	2.01
8 Motuihe	180.5	9	1.98
9 Rotoroa	89.2	21	1.92
10 Cuvier	**169.3**	**24**	**1.49**

Figure 2. The nine islands identified as high-priority sites for assisted seabird colonization, time since predator eradication and their GIS-MCDA scores in the Hauraki Gulf, New Zealand. The ellipses represent the 25 km distances from potential source populations. Restoration sites in bold are discussed as case studies. Cuvier Island (10) is discussed as an example that failed to meet the decision criteria, although has had limited recolonization and recovery of seabirds following the eradication in 1993 (Miskelly et al. 2009).

in large enough numbers on their own, it is possible that in the short term, predator eradications may not meet the ultimate restoration goal of ecosystem recovery (Kappes & Jones 2014). This study illustrates how spatially explicit ecological variables and predicted patterns of passive seabird recolonization can be used to prioritize islands that have been cleared of introduced predators for active seabird management. Although short-term restoration goals may be met through predator eradication alone, a GIS-MCDA approach can help inform an adaptive management approach for achieving long-term restoration goals. Our regression analysis and GIS-MCDA identified islands that had been cleared of predators and exhibited less than the expected species richness compared with islands of similar size without predators and islands beyond the probable natural dispersal distance from source populations (Buxton et al. 2014; Fig. 2). Such prioritization tools will be useful for island restoration or threatened species practitioners to help them decide whether active management of seabirds should be paired with predator eradication (Kappes & Jones 2014). Moreover, the GIS-MCDA framework is flexible and allows managers to choose criteria based on island and management specifics such as time, funding, logistics, species-specific conservation needs, stakeholder concerns, and/or restoration goals.

Defining restoration goals for seabird islands after predator eradication is not straightforward. Goals can be any combination of restoring the ecosystem engineering effects of seabird colonies, re-instating seabird assemblages similar to a pre-disturbed state or a comparable reference site, threatened species management, re-instatement of a cultural harvest of seabird chicks to restore socioecological systems (Towns 2002). For example, the Korapuki Island restoration plan aims to restore multiple components simultaneously, including vegetation, reptile, seabird, and invertebrate assemblages, in addition to providing a scientific testing ground for better understanding ecological restoration of seabird-driven systems (Towns & Atkinson 2004). Determining which goal is best for an island is based on the value judgements of a diversity of stakeholders, often creating obstacles to overcome (Suding 2011; Kappes & Jones 2014). If seabird recovery or ecosystem function is one of those goals, decision support tools can help stakeholders decide if active seabird management should be considered. The use of GIS-MCDA is illustrated with three examples of islands that have had predators removed spanning 21 years and with the highest GIS-MCDA scores (Table 1): Rakitu Island, which is soon to have ship rats (*R. rattus*) eradicated; Hauturu (Little Barrier) Island, which had Pacific rats and cats until their eradication was completed in 2004 (Veitch 2001); Cuvier Island, which was not identified as a high-priority site, having little observed seabird recovery since the eradication of Pacific rats in 1993 (Miskelly et al. 2009).

Rakitu Island was identified in our analysis as a high-priority site for assisted seabird colonization (Table 1, Fig. 2) because although there is suitable habitat and two seabird species present (fairy prions [*Pachyptila turtur*] and gray-faced petrels), potential source populations of Cook's petrels (*Pterodroma cookii*) and fluttering shearwaters (*Puffinus gavia*) are greater than 35 km away on Hauturu (Fig. 2). Furthermore, emigration from these source populations may be limited because there is habitat available for colony expansion on Hauturu itself, and thus density-dependent emigration is unlikely (Kildaw et al. 2005). Because there are no comparable reference sites and no complete history of Rakitu's fauna, the functional attributes of seabird species that may have been present on Rakitu prior to the invasion of rats are unknown. Although eradication is likely to be beneficial for the remnant seabird colonies, defining the restoration goal for Rakitu will be challenging, given differences in various stakeholder interests. For example, North Island weka (*Gallirallus australis*), a native species, were introduced to Rakitu and depredate small burrowing seabirds (Towns et al. 2011). The Weka Recovery Group would like weka to remain on Rakitu (Wilson 2013); however, their presence likely would counteract active seabird recovery efforts. Navigating complex stakeholder dynamics to come to a decision on restoration goals for any island restoration project warrants careful consideration. Engagement and discussion between stakeholders could be facilitated in the context of GIS-MCDA seabird recovery recommendations so that optimal, social, and ecological outcomes can be agreed upon before resources are allocated to assisted colonization of seabirds, or other resource-demanding actions (Suding 2011).

Conversely, environmental factors may play a larger role in defining restoration goals for seabird islands than reconciling stakeholder interests. For example, island area may inhibit the effects of seabirds because the physical and chemical influence of seabirds diminishes as island area increases (Ellis et al. 2011). There may be a maximum island area where seabirds can affect chemical and physical properties of the ecosystem, and thus modify or change the biotic communities of island habitats (Polis & Hurd 1995; Ellis et al. 2011). We identified Hauturu, the largest island included in this study, with the highest equal MCDA score (2.97) as a high-priority site for assisted colonization (Table 1; Fig. 2). The island is relatively depauperate in seabird species, given our species richness by island area analysis (Fig. 1), with only five species of the 14 Procellariiformes breeding in the region. Three are highly localized: the New Zealand storm petrel is only known from Hauturu, and small numbers of black petrels (*Procellaria parkinsoni*), and Cook's petrels are each known only from Hauturu and Great Barrier Island in the Hauraki Gulf.

Elucidating the extent of seabird engineering effects or predicting species–area relationships or assemblages of seabirds on Hauturu is challenging because there is no comparable reference site and relationships between seabird nutrient inputs on ecosystem function and island area are not well understood. However, the extent of available habitat means the island could theoretically support all seabird species included in our analysis (Fig. 1). Alternatively, Hauturu could be viewed as a refuge for threatened seabird taxa such as the three resident localized species with additions from those experiencing disproportional population trend declines (e.g. flesh-footed shearwater [*Puffinus carneipes*]; IUCN 2013). In this scenario, assisted colonization of threatened seabird species would be encouraged. Furthermore, although the recovery of seabird colonies on Hauturu is still in the early stages, we postulate that it could serve as an important seabird source for other islands in the region in the future, acting as a biodiversity "insurance policy," by enhancing local populations of species that may not be listed as threatened. However, this is contingent on the passive or assisted colonization of seabird taxa currently absent from Hauturu.

Our GIS-MCDA analysis did not identify Cuvier Island as a high-priority site for assisted seabird colonization because it lies on the edge of the 25 km radius of source populations (Fig. 2). However, it has not had any passive recolonization of seabirds since the eradication of Pacific rats in 1993 (Miskelly et al. 2009). Although passive recovery may still happen given more time (Jones 2010*b*), this example highlights the need for users to have a comprehensive understanding of the target islands and the limitation of GIS-MCDA, where islands close to the "cut-off" values used may need to be examined on a case-by-case basis. Cuvier Island is known to have small colonies of gray-faced petrels, common diving petrels (*Pelecanoides urinatrix*), and fluttering shearwaters (Miskelly et al. 2009). One additional species, Pycroft's petrel (*Pterodroma pycrofti*), was used as an experimental translocation to the island (G. Taylor 2014, Department of Conservation, personal communication). The available habitats on Cuvier Island could support up to 10 seabird taxa (Fig. 1). There are sufficient source populations of species such as little shearwaters (*Puffinus assimilis*) and flesh-footed shearwaters from the Mercury Island group, which are approximately 20–24 km from Cuvier Island (Fig. 2). Environmental conditions may be suitable on Cuvier Island, but complex interactions between resource distributions, heterospecific competition, and metapopulation dynamics (Oro 2003) may inhibit the ability of the target species to recolonize or maintain a viable population despite the proximity of source populations. Currently, there is limited understanding of metapopulation connectivity and dynamics among seabird colonies, and future work in this area will contribute to enhancing decision tools for restoration managers.

Globally, nearly one third of seabird species are threatened with extinction (IUCN 2013). Therefore, restoration actions that enhance threatened seabird species or facilitate the establishment of new breeding colonies are important for achieving international goals of enhancing and sustaining the world's seabirds (Birdlife International 2013). Furthermore, seabirds are ecological and cultural keystone species, and their recovery is integral to the restoration of socioecological seabird island systems (Buxton unpublished data). If islands are to be restored to seabird-driven ecosystems comparable with a suitable reference site, then attracting seabirds back to those islands cleared of introduced predators may be necessary in the absence of successful passive recolonization (Kappes & Jones 2014). GIS-MCDA that combines habitat suitability and drivers of passive seabird recolonization can be an informative tool for conservation managers to direct limited conservation resources in a manner that is most beneficial for ecosystem recovery, and for single-species conservation alike. We do not suggest that there is little value in restoring islands ranked as low-priority sites for active seabird management. Indeed, there may be considerable conservation benefits through restoring these islands associated with restoration goals that are beyond the scope of this paper.

We believe that a GIS-MCDA approach can effectively prioritize island restoration sites for assisted colonization of seabirds to islands. Such an approach could aid managers to revisit their restoration goals. For example, based on the decision output and available resources, should the goal be to re-instate system process, restore species assemblages, manage threatened species, or are there multiple goals for the island? Ultimately, seabird island restoration projects should aim to promote or enhance seabird-driven processes and the sustainability of viable populations that are resilient to unpredictable ecological disturbances inflicted by climate change or other factors (Margules & Pressey 2000). Regardless of the decision to focus on functional drivers or species assemblages, managers must take an informed and active adaptive approach. GIS-MCDA could provide a valuable tool to integrate the social, economic, and ecological factors to inform island restoration actions.

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Supporting Information

The following information may be found in the online version of this article:

Appendix S1. Detailed methods and data tables.

Appendix S2. Island area, year eradicated of introduced predators and population estimates of the islands included in the GIS-MCDA for prioritizing active seabird management on islands in the Hauraki Gulf, New Zealand.

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