

Research Article

Reinstatement of Customary Seabird Harvests After a 50-Year Moratorium

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ABSTRACT Seabird eggs, chicks, and adults have significant value for many cultures, but delayed maturation, low reproductive rates, and ease of exploitation at breeding colonies make these species especially vulnerable to overharvest. In New Zealand, indigenous Māori placed a moratorium over the harvest of grey-faced petrel (*Pterodroma gouldi*) chicks in the 1960s because of concerns about diminished returns. Over the last decade, those tribes have indicated their desire to reinstate a small customary harvest. We used recent estimates of population sizes and demographic parameters to develop population models to investigate the feasibility of reinstating a sustainable harvest of pre-fledgling chicks from 2 island populations: Moutohorā and the Ruamāhua Islands. In the absence of harvest, our simulations predicted mean annual rates of population growth of 1.01 (95% CI 1.010–1.014) for the colony on Moutohorā and 0.983 (0.978–0.989) for the Ruamāhua Islands. The Moutohorā colony could potentially sustain a fixed-quota harvest of up to 6,000 chicks or a fixed-proportion harvest of up to 30% of chicks, annually. A larger-scale harvest would require either a very conservative fixed-quota harvest with strict monitoring of bag limits, or a fixed-proportion harvest with reliable ongoing monitoring of the population size. We propose that Māori authorities use a fixed-quota system up until they develop and use either a scientific assessment or a harvester-derived index of population change. On the basis of our simulations, in 2010, Ngāti Awa, the local tribe, harvested chicks from Moutohorā for the first time in half a century. © 2014 The Wildlife Society.

KEY WORDS grey-faced petrel, harvest, indigenous culture, Māori, modeling, New Zealand, *Pterodroma gouldi*, seabirds, sustainability.

Harvests of seabird eggs, chicks, and adults occur globally (Skira et al. 1986, Conservation of Arctic Flora and Fauna Working Group of the Arctic Council 2001, Baker et al. 2004). Colonial burrow-nesting petrels (Order: *Procellariiformes*) in particular have significant nutritional, cultural, economic, and biological value for many cultures (Skira 1996, Lyver et al. 2008). Delayed maturation, low reproductive rates, and ease of exploitation at breeding colonies make these species especially vulnerable to overharvest (Croxall et al. 1984, Moller 2006). The effects of overharvest, combined with those of degradation of breeding habitat, predation by introduced predators (particularly on islands), losses as by-catch in fisheries, and environmental pollution, have contributed to declines in breeding populations worldwide (Croxall et al. 2012). These effects have often resulted in restrictions on harvests through protective legislation, harvest prohibition (using both legislative and indigenous strategies), and land reservation

(Moller 2006). For communities and indigenous peoples, this can lead to a loss of values (and well-being), connection to culture, harvest practices, and long-term knowledge about the bird and its environment (Berkes 1999, Tau 2001, Moller et al. 2004, Lyver et al. 2008).

Modern scientific management of sustainable wildlife harvests is based on using empirical estimates of population size and demographic rates, which, in turn, are used to predict levels of off-take that can be achieved without imperiling the long-term viability of the harvested population. Harvest limits are most commonly set as fixed quotas of individuals, fixed proportions of the population, or constrained to occur only when the population is greater than some threshold size (Fryxell et al. 2005, Bradshaw et al. 2006). Predicting the impacts of management interventions or harvest on seabird populations presents a particular set of problems because of their characteristically long generation times, through delayed maturity and low recruitment rates (Croxall and Rothery 1991). These life-history rates can also be difficult to measure. This means that any resulting changes in growth rates of these populations are likely to take several years to be expressed and detected (Croxall and Rothery 1991, Russell 1999).

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The use of tactical mathematical simulation models can help to identify the most influential factors affecting population growth and allow probabilistic predictions of the relative impacts of various management options (Starfield and Bleloch 1986, Kareiva and Anderson 1988, Starfield 1997). Models incorporating age- or life-history stage-structure are particularly useful in predicting the effects of removing particular demographic groups from a population, which occurs frequently in seabird harvests (McDonald and Caswell 1993, Hunter and Caswell 2005, Mills 2013).

In New Zealand, chicks of the grey-faced petrel (*Pterodroma gouldi*) were harvested traditionally from islands off the northeast coast by northern iwi (tribes) of indigenous Māori. Hauraki and Ngāti Awa are 2 such iwi, who, in the mid to late 1960s, implemented rāhui (harvest moratoria) because of concerns about diminishing harvest returns of chicks (Imber 1976, Lyver et al. 2008). Because of the great cultural, social, and spiritual significance of the traditional practices associated with harvest, the iwi have expressed interest recently in whether the colonies had recovered enough to support customary harvest on a small scale. The communities were also fearful that their indigenous knowledge, practices, tribal identity, and associated values would be lost if the moratoria remained for another generation. There are also increasing numbers of requests from conservation organizations to remove chicks from these islands for translocation and restoration purposes elsewhere. In such situations, where the skills and management concepts of traditional harvesting have been eroded because of restrictions on harvesting, scientific methodologies, including the use of predictive modeling, can help to guide wildlife managers and indigenous groups on safe and sustainable levels of off-take (Heinsohn et al. 2004, Bradshaw and Brook 2007, Gorman et al. 2008).

We used recently estimated demographic parameters for grey-faced petrels in tactical population models to explore the effects of 2 harvest strategies—1 based on removing fixed quotas and the other on removing fixed proportions of chicks annually—on the petrel populations of 2 island systems. We used predictions from these models to advise both the iwi and a government wildlife management agency about the sustainability of reinstating harvests of chicks from these populations.

STUDY AREA

Our study populations were on the 4 largest islands within the Ruamāhua (Aldermen) Islands archipelago (36°57' S, 176°50' E, combined area approx. 100 ha) and the single island of Moutohorā (Whale Island; 36°51' S, 176°58' E, 173 ha; Fig. 1). Soils on the islands were volcanic, and were characteristically well drained but quite friable (Hayward and Moore 1973, Rijkse 1980). The climate for both study sites was warm-temperate; the mean annual rainfall and temperature on the mainland nearest to the Ruamāhua Islands were 1,840 mm and 14.7°C, respectively, and for Moutohorā, 1,189 mm and 14.0°C, respectively (National Institute for Water and Atmosphere 2012).

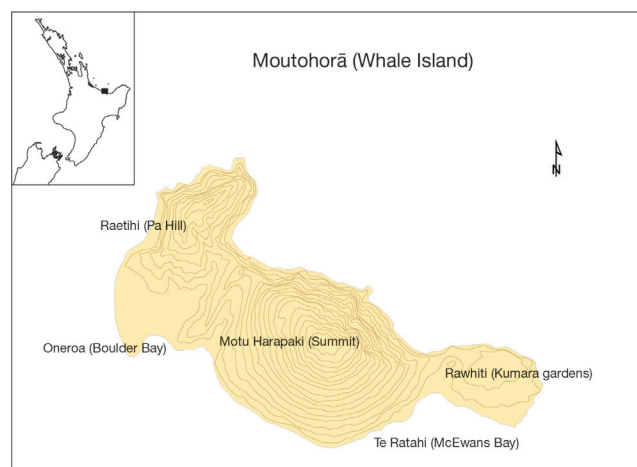
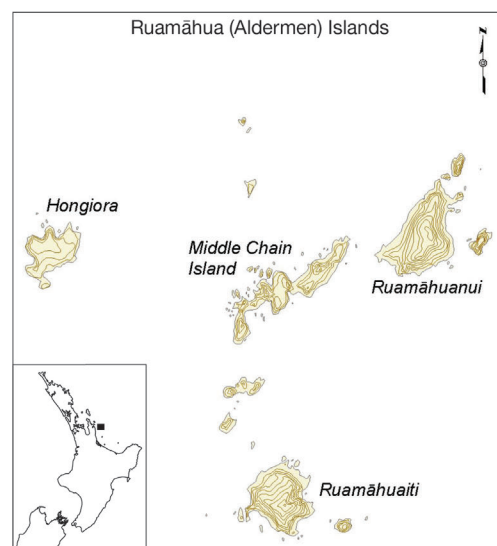


Figure 1. Map of the Ruamāhua (Aldermen) Islands and Moutohorā, home to modeled grey-faced petrel (*Pterodroma macroptera gouldi*) populations. Inserts indicate island locations relative to the North Island of New Zealand.

The vegetation of the larger islands of the Ruamāhua group and Moutohorā consisted of mixed forest dominated by angiosperm trees. Forests on all the islands were cleared initially by Māori, but only Moutohorā was farmed by Europeans. Forests have been regenerating naturally on the Ruamāhua Islands since 1935 (Sladden and Falla 1927, Cochrane 1962, Court et al. 1973). During our study, the vegetation was generally less mature on Moutohorā resulting from extensive replanting between 1984 and 1989 when it was purchased and accorded reserve status by the government (Department of Conservation 1999). Recent estimates of the numbers of breeding pairs of grey-faced petrel were similar for both systems, with 72,410 (95% credible interval 14,280–138,400) estimated breeding across the Ruamāhua Islands and 69,330 (10,590–128,300) on Moutohorā (Whitehead et al. 2014).

The 4 Ruamāhua Islands supported populations of several burrowing seabirds, including grey-faced petrels, fluttering shearwaters (*Puffinus gavia*), little shearwaters (*P. assimilis*),

common diving-petrels (*Pelecanoides urinatrix*), little blue penguins (*Eudyptula minor*), and white-faced storm-petrels (*Pelagodroma marina*; on Hongiora only). Pacific rats (kiore [*Rattus exulans*]) were present on 1 island in the Ruamāhua archipelago, Middle Chain, from which they were eradicated in 1994 (Clout and Russell 2006) otherwise, no introduced mammalian predators have been recorded on the islands of that group.

METHODS

Study Species and Populations

The grey-faced petrel is a gadfly petrel (average adult body mass 550 g) that breeds on headlands and islands primarily off the east coast of northern New Zealand (Heather and Robertson 2005). The species is a winter breeder that lays a single egg in burrow nests towards the end of June and into July. Hatching occurs from mid-August to late September with chicks fledging in late December and early January (Imber 1976). The population of grey-faced petrels in New Zealand has been estimated to total 1,000,000 breeding and nonbreeding individuals (Taylor 2000). Although previously considered a sub-species of the great-winged petrel (*Pterodroma macroptera*; Marchant and Higgins 1990), the 2 are now considered separate (del Hoyo et al. 2014).

Grey-faced petrel chicks were harvested traditionally from Moutohorā by Ngāti Awa with accounts of >50,000 chicks harvested in a single breeding season (van der Wouten 1994). Harvests continued up until the early 1960s, when a rāhui was put in place because of concerns about diminishing returns. Moutohorā was then removed from the list of breeding islands for which the New Zealand government allowed harvest permits to be issued. The primary cause for

this decline is not known, but Norway rats (*Rattus norvegicus*) were present on the island from approximately 1920, with their population boosted by the presence of rabbits (*Oryctolagus cuniculus*) from the mid-1960s to such an extent that few petrel chicks fledged at all between 1972 and 1977 (Imber et al. 2000). Domestic cats associated with European occupation of the island were last recorded there in the 1960s; sheep, cattle, and goats were removed from the island in the mid-1970s, and rabbits and Norway rats were eradicated in 1986 (Harrison 1992, van der Wouden 1994, Imber et al. 2003). Historically, other species of petrel have bred on Moutohorā, but the island now supports only grey-faced petrels, and a few little blue penguins around the coast (Imber et al. 2000).

Harvest Modeling

We constructed an age-structured matrix model (Leslie 1945, McDonald and Caswell 1993, Caswell 2001) to describe the growth of the grey-faced petrel populations in each of the 2 island groups. We considered the 4 Ruamāhua islands to represent a single population because no other breeding population of significant size (>5,000 breeding pairs) occurs within 40 km in any direction (Taylor 2000). We based the modeled populations on a simplified life cycle for the grey-faced petrel (Fig. 2).

The modeled population is sampled when chicks fledge each year. The number of eggs laid each year depends on the number of adult birds and the proportion of these that breed. There are no published estimates of the proportion of breeding-age adult grey-faced petrels breeding in any 1 year, so we used the mean of the rates for great-winged petrels that had bred successfully and unsuccessfully in the previous

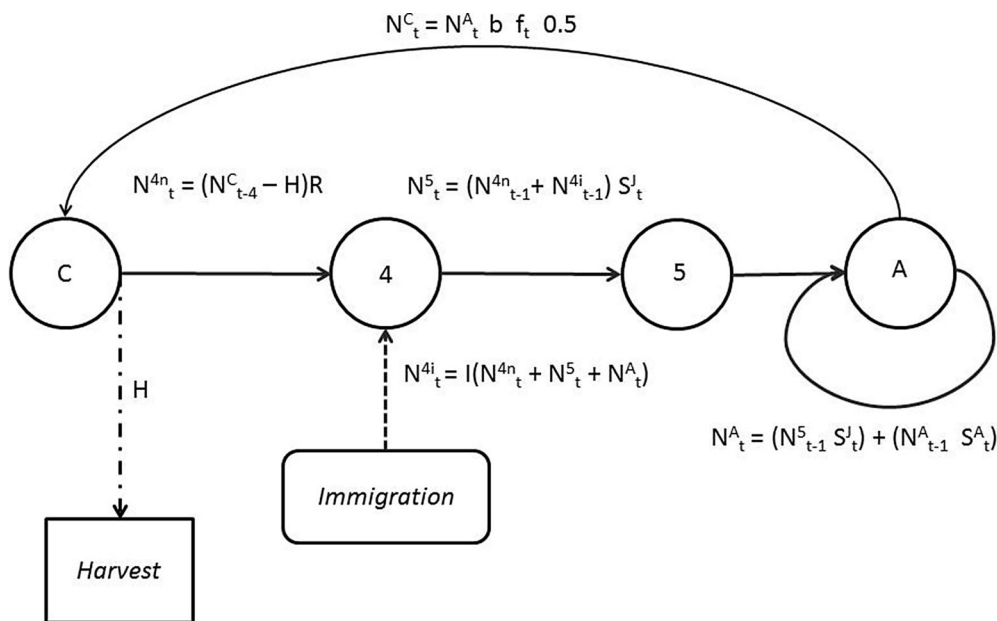


Figure 2. Life-cycle diagram summarizing post-breeding census model for grey-faced petrels subject to harvest. Numbers of individuals (N) in each age class (denoted by the following superscripts: chicks, C; 4- and 5-year-old pre-breeders, 4 and 5, respectively; adults, A) vary annually according to the following parameters: annual adult survival (S^A), annual pre-breeder survival (S^J), breeding success (f), proportion of adults breeding (b), probability of return to the natal colony at age 4 years (R), and immigration rate (I). Returning pre-breeders are denoted by 4^n and immigrants by 4^i . Harvest (H) occurs immediately prior to fledging as either a fixed-quota or a proportion of the available chicks. Time-varying parameters are denoted by the subscript, t .

season (Chastel 1995). The latter are closely related to grey-faced petrels and have similar life-history characteristics. Very similar annual rates of participation in breeding have been recorded for other procellariiform species (Mougin et al. 1997, Bradley et al. 2000).

In the absence of estimates for the probability of breeding with age for this species, we made the simplifying assumptions that 4- and 5-year-old birds are pre-breeders and had no reproductive output and that birds aged 6 and over were considered breeding adults (Marchant and Higgins 1990; G. Taylor, New Zealand Department of Conservation, personal communication). Although there is likely to be variation in individuals' probability of transition to breeding with age, investigations of the effects of simplifying age structure in demographic models have demonstrated that this has little effect on estimates of population growth rates or on perturbation analyses (Heppel et al. 2000, Hunter et al. 2000).

We modeled the probability that an egg produces a fledgling according to annual island-specific breeding success rates estimated for both modeled populations by Jones et al. (2014) using logistic exposure modeling of burrow survey data from 2006 to 2010. We included both male and female birds and assumed the sex ratio of young to be equal, as is the norm in procellariiform seabirds (Genovart et al. 2005, Lawrence et al. 2008, Medeiros et al. 2012).

After fledging, juvenile birds leave the colony and return, on average, at 4 years of age. We included apparent survival between fledging and return at age 4 in the model as a probability of return at this point using estimates based on a recent mark-recapture analysis of 18 years of grey-faced petrel banding data from a northern New Zealand population (Jones et al. 2011). We based estimates of mean adult and 4- and 5-year-old pre-breeder annual survival rates and standard deviations around these on the mark-recapture analyses of Jones et al. (2011), and C. Jones (Landcare Research, unpublished data).

We assumed immigration of new individuals occurred as an increase in the number of 4-year-old birds establishing at the colony and fixed it at 7% of the total population in each year. This was based on published estimates for short-tailed shearwaters (*Puffinus tenuirostris*; Serventy and Curry 1984, Bradley et al. 1991, Hunter et al. 2000) and is supported by other published estimates for procellariiform seabirds (Chastel et al. 1993; Jenouvrier et al. 2003, 2005; Oro et al. 2004; Peery et al. 2008).

We estimated the starting population size as the sum of 1) the number of adult birds, estimated from the number of breeding pairs in each population (Whitehead et al. 2014) corrected by the proportion of the adult population that breeds in any 1 year, and 2) the numbers of 4- and 5-year-old birds in the starting population, which were estimated as fixed proportions of the adult population using the stable age-distribution in a preliminary deterministic model of each population. We incorporated environmental stochasticity by sampling adult survival and breeding success randomly from β -distributions defined by their means and standard deviations using the Poptools version 3.2.3 add-in for Excel (<http://www.poptools.org>; Table 1). We made both parameters co-vary using a common random root value to simulate years of good and bad environmental conditions (White 2000).

We did not include any density dependence in our model for 3 reasons. First, only about half of the available breeding burrows on the island were occupied by breeding pairs during our survey period, which would suggest that the population is far from the carrying capacity of its breeding environment. Second, although various mechanisms have been postulated to account for density-driven effects on vital rates in seabirds (Lewis et al. 2001, Tavecchia et al. 2007, Jones et al. 2014), we have little specific information on which to define an effect. Third, oral history accounts from Māori elders suggest that breeding populations on our study islands have declined dramatically since the early 20th century (Lyver et al. 2008).

Table 1. Parameter values used in models to estimate population size for grey-faced-petrels (*Pterodroma gouldi*) on offshore islands of New Zealand. All parameter values are based on published estimates for the focal species. Where these were unavailable, we used appropriate estimates from other procellariiforms: a) great-winged petrel (*Pterodroma macroptera*) and, b) short-tailed shearwater (*Puffinus tenuirostris*). Values in parentheses are standard deviations used in simulating environmental stochasticity.

Parameter	Estimate (SD)	Source
Adult (≥ 6 yr old) annual survival	0.89 (0.01)	Jones et al. (2011), C. J. Jones, Landcare Research, unpublished data
Pre-breeder (4–5 yr old) annual survival	0.89 (0.01)	Jones et al. (2011), C. J. Jones, unpublished data
Age of first return (AFR) to natal colony	4	Jones et al. (2011)
Probability of return at AFR	0.58	Jones et al. (2011)
Age at first breeding	6	Marchant and Higgins (1990), G. Taylor, New Zealand Department of Conservation, personal communication
Probability of adults breeding	0.8605 ^a	Chastel (1995)
Breeding success	0.280 (0.04; Moutohorā) 0.127 (0.02; Ruamāhua Is.)	Jones et al. (2014)
Immigration parameter	0.07 ^b	Bradley et al. (1981), Serventy and Curry (1984), Hunter et al. (2000)

We simulated harvest as the removal of fledglings before they leave the colony. We simulated 2 general strategies: 1) fixed-quota harvest, where a set number of chicks, ranging from 200 to 15,000 are removed every year, and 2) fixed-proportion harvest, where 1–60% of chicks are removed every year. We simulated changes in the population for 50 years following the implementation of harvest. We simulated each harvest strategy 500 times using the Monte-Carlo simulation algorithm in Poptools 3.2.3, from which we derived a mean population growth rate, λ .

To assess the relative risk to the breeding population due to each harvest strategy, we estimated the expected minimum population size as the mean of the smallest population size from each iteration of the Monte Carlo simulation for each scenario (McCarthy 1996, McCarthy and Thompson 2001). We expressed this risk as the expected population decline, i.e., the percentage decrease from the starting population size that the expected minimum population size represents (Fordham et al. 2008, Firth et al. 2010). The overarching research from which we obtained much of the data in this paper had ethics approval from the Landcare Research Animal Ethics Committee (Permit Nos 06/02/03 and 10/03/05).

RESULTS

In the absence of any harvest, our simulations predicted a mean annual population growth rate, λ , of 1.012 (95% CI 1.010–1.014) over 50 years for the grey-faced petrel population on Moutohorā. In contrast, mean λ for the Ruamāhua Islands population was 0.983 (0.978–0.989), suggesting that the population is declining and is therefore unable to sustain any harvest of chicks. We therefore restricted our subsequent assessments of the effects of harvest strategies to the Moutohorā population.

Under a fixed-harvest quota strategy, we found the Moutohorā population was able to sustain an annual harvest of up to 6,000 chicks, beyond which estimates of expected population decline increased rapidly (Fig. 3a). Harvests of above 10,000 chicks per year led to scenarios where eventually no chicks fledged, the population consisting of slowly decaying numbers of adults and, accordingly, increasingly fewer immigrants.

A fixed-proportion harvest of up to 30% of chicks was sustainable (Fig. 3b). The buffering effect of proportional off-take on the absolute numbers of chicks taken resulted in 2 distinct differences from a fixed-quota harvest. First, expected population declines were less dramatic than those predicted from fixed-quota strategies. Second, the situation where no chicks fledged was never reached.

DISCUSSION

Fixed-Quota Versus Fixed-Proportion Harvest Strategies

Under a fixed-quota strategy, our simulations predicted a low risk of decline on Moutohorā until at least 6,000 chicks were taken annually, so this approach would be appropriate for the management of a small, largely symbolic harvest aimed predominantly at maintaining harvesters' cultural and

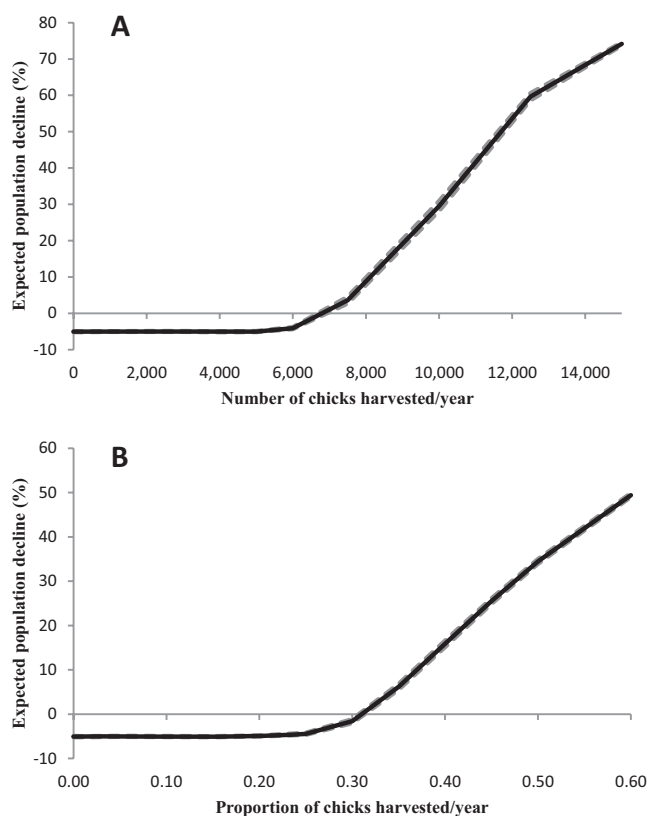


Figure 3. Predicted effects of varying the intensity of A) fixed-quota, and B) fixed-proportion harvest of chicks on expected population decline of the grey-faced petrel breeding population on Moutohorā, Bay of Plenty, New Zealand. Dashed lines are 95% confidence intervals.

spiritual links to the resource. These predictions are based on the assumption that grey-faced petrel demographic parameters remain within the bounds of the sampling distributions used in the model. If a large-scale harvest was to be re-established, the fixed-quota strategy would be risky because it does not make an allowance for any compensatory reduction in harvest should significant environmental variation compromise 1 or more demographic rates (Bradshaw et al. 2006). On the other hand, should environmental change benefit the breeding population, a fixed-quota strategy can also lead to a loss of harvest opportunities (Hunter and Runge 2004).

Using a fixed-proportion harvest strategy, up to 30% of chicks could be harvested before the population was predicted to decline. Clearly, this approach is safer in a stochastic environment as harvest is scaled according to the availability of the resource. Thus, if numbers of grey-faced petrels decline because of other nonharvest impacts, compensatory adjustments in harvest would serve to mitigate these effects on the population to some degree. Notably, reducing harvest may be insufficient to mitigate population declines if the impacts are on those demographic classes within the population possessing high reproductive value; long-lived seabird populations are particularly vulnerable to losses of breeding-age adults (Hunter et al. 2000, Moller 2006). The main drawback of a fixed-proportion harvest strategy is that it requires on-going precise and accurate assessment of the harvest population to allow

informed adjustment of harvest limits (Fryxell et al. 2005, Bradshaw et al. 2006).

Our estimates of safe harvest limits were based on stochastic projections but did not incorporate uncertainty in the breeding population sizes for our modeled populations. We obtained estimates of breeding population sizes using complex habitat-area models (Whitehead et al. 2014), which, essentially, sacrifice precision for accuracy compared with more simplistic approaches to population estimation (Rayner et al. 2007). In situations where models are used to guide the management of critically imperiled species or populations, a highly precautionary approach would be to use a lower bound on the population estimate to define a starting population. For relatively more abundant species and populations, using estimated population decline as an indicator of the relative risk allows impacts on a population to be estimated across a range of population sizes and, unlike other published metrics, can indicate the likelihood of decline even when the risk of extinction is relatively minor (McCarthy et al. 2003).

Differences in Population Growth Rates on Moutohorā and the Ruamāhua Islands

In contrast to the grey-faced petrel population on Moutohorā, our models indicated that the Ruamāhua Islands' population is declining. This corresponds with reports by Māori harvesters of dramatic declines in catch rates on the islands, particularly since the 1980s (Lyver et al. 2008). Given that harvesters are likely to target the most productive areas of breeding colonies to compensate for this overall reduction in chick availability, the reported declines may be underestimates of true population changes (Lyver 2002, Moller et al. 2004). Taken together, published evidence from harvester interviews and our simulations suggest that any level of harvest should be avoided on the Ruamāhua Islands.

Both of our study populations are of similar size (approx. 69,000 breeding pairs on Moutohorā and 72,000 on the Ruamāhua Islands; Whitehead et al. 2014) and are likely to be subject to similar mortality risks and forage in similar oceanic environments. This suggests that local factors may be driving the contrasting trajectories of the 2 populations. Clearly, the biggest difference in our input parameters for the 2 populations was in the estimates of breeding success (13% on the Ruamāhua Islands vs. 29% on Moutohorā). In estimating these rates, Jones et al. (2014) proposed 2 potential mechanisms for this difference on the basis of the different characteristics of the 2 island ecosystems. Our study systems reflect different points of a recovery–restoration trajectory following the cessation of human use (including burning and agriculture) and the impacts of invasive rats. The Ruamāhua Islands never had Norway rats present, but this species was on Moutohorā until relatively recently. Furthermore, the native forest on the Ruamāhua Islands is at least half a century older than that on Moutohorā. Differences in canopy maturity and density may drive differences in productivity by limiting the ease and safety of access to the forest floor for returning breeders; several

studies have suggested that higher procellariiform densities are associated with lower and less dense forest canopies (Rayner et al. 2007, Scott et al. 2009, Whitehead et al. 2014) as found on Moutohorā. In addition to having generally higher and denser forest canopies, the Ruamāhua Islands also support breeding populations of a range of other burrow-nesting procellariiform seabirds and there is evidence that grey-faced petrels are sensitive to disturbance during the breeding season, leading to nest desertion (Imber 1976, Johnstone and Davis 1990, Cary 2011).

On the basis of our preliminary modeling in 2010, the New Zealand Minister for Conservation approved the re-establishment of a small-scale customary harvest of 200 chicks on Moutohorā, with a further 50 chicks permitted to be removed for translocation. Ngāti Awa, in December 2012, under the regulatory guidance of the Te Tapatoru a Toi (Moutohorā Joint Management Committee), harvested approximately 140 grey-faced-petrel chicks from Moutohorā for the first time in half a century. The reinstatement of the harvest of grey-faced petrels on Moutohorā was a long-held vision of Ngāti Awa elders and represented the return of a significant part of their people's identity and mana (prestige).

MANAGEMENT IMPLICATIONS

If the decision is ever taken to instigate a larger-scale harvest on Moutohorā than the current limit of 250 chicks, 2 options exist for regulating the number of chicks taken each year. The first strategy would be to set a very conservative fixed-quota harvest with strict monitoring of bag limits. The second option of setting a fixed-proportion harvest strategy is less risky, but because it requires on-going monitoring of the population size, monitoring must be reliable enough to track real changes in the population. This would require either continued input from wildlife professionals, which is expensive to maintain in the long term, or would require the local community to be trained in and to take responsibility for tracking population changes. Much of the mātauranga (Māori traditional knowledge) associated with harvest of chicks from Moutohorā has eroded with the passing of those older members of the iwi who participated in the annual intensive harvesting before the moratorium. With the erosion of this knowledge, community-based indices for monitoring, requiring experienced assessment are unlikely to be feasible to use in the short term. Therefore, using a science-based index in predefined and permanent monitoring areas may be appropriate.

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