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Technical Report 176

**Newell's shearwater population modeling
for Habitat Conservation Plan and Recovery Planning**

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Pacific Cooperative Studies Unit
&
State of Hawaii
Department of Land and Natural Resources
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Summary

The Newell's shearwater (*Puffinus auricularis newelli*), an IUCN and ESA listed species, faces terrestrial threats from predation, fallout (attraction to artificial lights) and collision with powerlines. Various indices suggest the population has declined by ~75% in the past two decades. Population modeling is required for Habitat Conservation Plan (HCP) and Recovery Planning to consider the benefits of existing and proposed management actions to the Kauai population.

Population scenarios modeled here included a) stable, realistic and optimal growth; b) threats of predation, fallout and powerline collision; and c) management actions of minimizing fallout and powerline mortality, the Save Our Shearwater rescue program, predator control, predator eradication and chick translocation. The growth rate (λ) produced in our worst case threat scenario for all threats (0.906) fell within the range of annual change suggested by ornithological radar data from 1993-2010 using only Newell's shearwater traffic (0.899), and Save Our Shearwater data of Newell's shearwater fledglings from 1988-2009 (0.905).

When considered independently, fallout and powerline minimization have the potential to increase growth rate by up to 0.5% and 0.3%, respectively, but would prove more effective in areas where concurrent colony management is planned. The Save Our Shearwater program could theoretically increase growth rate by up to 0.8% if fledglings recovered and released experience the same survival as fledglings' not experiencing fallout. While experiencing powerline strike and fallout threats, predator control with 90% effectiveness produced population growth rates from 0.976 to 0.991 and predator eradication produced growth rates from 0.986 to 1.00, suggesting additional growth of 0.2% - 2.4% would be required to stop a decline in these areas. We allowed translocated populations to grow at 1.012 because chicks would likely only be moved to areas free of terrestrial threats.

Combined management scenarios with 1) 10% population subjected to predator control, 5% to predator eradication, and 100 chicks translocated over five years, and 2) 20% population subjected to predator control, 10% to predator eradication, and 400 chicks translocated over 10 years, would provide a benefit of 2,000 and 4,000 birds over 25 years, respectively, compared to no management undertaken. The benefits from these actions have clear potential to offset incidental take proposed in HCP planning.

A recovered (i.e. stable, self-sustaining) Newell's shearwater population in 25 years will likely be much reduced in size from what exists today. However, it is certainly realistic that subsets of the population can achieve positive growth rates when predation, fallout and powerline mortality are removed. Combined management actions likely to provide the greatest potential benefit should be targeted to prevent this species from becoming extinct, including predator control of Northwest colonies, predator eradication projects in combination with aggressive powerline and fallout minimization in the same region, and chick translocation to threat-free environments.

Keywords: Newell's shearwater, petrels, endangered, population modeling, light attraction, powerline collision, predation, management, Save Our Shearwaters, minimization, predator control, predator eradication, chick translocation

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1 Introduction

The Newell's shearwater *Puffinus auricularis newelli* is an endemic Hawaiian seabird listed as Threatened under the US Endangered Species Act (ESA) and endangered on the International Union for the Conservation of Nature (IUCN) Red list (IUCN 2010, PL 93-205). Newell's shearwaters are nocturnal, burrow and crevice nesting, colonial breeding petrels (Procellariiformes), with key life history characteristics reflecting a k-selected demographic strategy, delayed breeding, low fecundity, high adult survival and only returning to land to breed (Warham 1990, Warham 1996).

As for so many seabirds that evolved on islands, Newell's shearwaters are entirely naïve to the threats posed by predatory mammals or humans (Nettleship et al. 1994). Known terrestrial threats include predation (non-native cats *Felis catus*, rats *Rattus* sp., barn owls *Tyto alba*), habitat modification from invasive plant species and ungulates (pigs *Sus scrofa*, goats *Capra hircus*), and attraction to artificial lights and collision with artificial structures (Ainley et al. 2001, Sincock and Swedberg 1969).

Population indices of Newell's shearwaters have declined precipitously in the past two decades. Based on at-sea observations from 1984-1993, their population was estimated at 84,000 birds, with ~90% nesting on Kauai (Spear et al. 1995). Between 1993 and 2009, population indices of ornithological radar, and the numbers of downed fledglings collected after attraction to artificial lights (fallout birds) declined by ~75% (Day et al. 2003, Holmes et al. 2009). Concurrently, a breeding range contraction appears to be occurring based on reduced activity at three Newell's shearwater breeding colonies known to be active between 1980-1994 (Holmes et al. 2009). At-sea threats are poorly understood and require investigation, however minimizing terrestrial threats is crucial to the recovery of this species.

Population demography models and Population Viability Analyses (PVA) are used by wildlife managers to inform decision making processes (Caswell 2001, Keedwell 2004). PVA methods are well established and can have a high degree of accuracy over a short-term period (i.e. 20-30 yrs). A comparison of five PVA software methods showed high short-term predictive accuracy when compared to observed population changes amongst 21 long-term studies of wildlife populations (Brook et al. 2000). For managers, PVA can be used to compare the impact of different threats, the efficacy of different management scenarios, and evaluate which life stages should be the focus of management efforts (Morris et al. 1999).

Previous population modeling of Newell's shearwaters were undertaken in 1995 (Ainley et al. 2001), when the dramatic decline in ornithological radar targets and fallout birds was not yet evident. Based on estimated life history parameters and measures of mortality from predation, light attraction, and collision with human-made structures, model outputs predicted annual declines of 3.2 to 6.1 % (λ of 0.968 to 0.939) or decadal declines of 30-60% (Ainley et al. 2001).

Light attraction and collision of Newell's shearwater with human-made structures is considered "incidental take"¹ under the ESA and Hawaii Revised Statutes Section 195D where impacts to the species

¹ The term "take" as defined by the ESA means to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct PL 93-205. 87 Stat. 884, 16 U.S.C. §§ 1531-44. Endangered Species Act of 1973.. Per HRS 195D-4, "Take" is defined as to: "harass, harm, pursue, hunt, shoot, wound, kill, trap,

are unavoidable. The Hawaii Division of Forestry and Wildlife (DOFAW) is addressing the incidental take through the development of the Kauai Seabird Habitat Conservation Plan (KSHCP). The KSHCP is an island-wide plan for multiple entities applying for authorization of incidental take via federal and state permits (KSHCP 2010). Hawaii state permit issuance criteria require that HCPs must provide a net benefit² to the species in addition to offsetting unavoidable incidental take. Biological actions to offset unavoidable take and provide net benefit as specified in the HCPs are defined as "mitigation". Proposed mitigation activities of the KSHCP include removing predators from breeding colonies and controlling habitat modifiers (KSHCP 2010).

The actions to minimize and address incidental take are part of a larger scope of recovery planning that addresses terrestrial and marine threats to this species' population (Holmes et al. 2011, USFWS 1983). Recovery planning requires determination of the most effective management strategy to stop the decline, and allow the population to grow.

The overall aims of the current paper were to guide KSHCP mitigation goals and recovery actions for Newell's shearwaters by a) describing life history, b) quantifying effects of threats on the population, c) comparing outcomes of threats on population to existing population indices of radar and fallout data, d) identifying potential outcomes of management actions, including facility based actions of minimizing light and powerline mortality, the Save Our Shearwaters (SOS) fallout rescue program, and colony based management actions of predator control, predator eradication and chick translocation. This document is for use by government agencies, land managers, researchers, and other stakeholders with technical expertise in conservation biology that are involved in HCP mitigation planning and/or Newell's shearwater recovery planning. Importantly, in regard to HCP planning these results should not be considered a comprehensive assessment of net benefit from ameliorating predation threats, but rather a starting point for selecting an appropriate mitigation investment. Investments should be considered tentative until HCP monitoring provides actual measures of productivity in management areas.

2 Methods

2.1 Modeling approach

For population modeling, we used stage-based Lefkovitch matrices, based on Caswell (2001) using PopTools v3.1 (Hood 2009) and RAMAS Metapop v5.0 (Akçakaya and Root 2007). Matrix modeling was based on life history parameters of survival and fecundity (age of first breeding, breeding probability, reproductive success and sub-adult and adult survivorship) (Figure 2-1 and Figure 2-2).

capture, or collect any threatened or endangered species, including plants, animals, birds, fresh and marine water species." Incidental take occurs unintentionally and incidental to the use of otherwise lawful facilities.

² HRS §195D-30: "All habitat conservation plans, safe harbor agreements, incidental take licenses, and subsequent actions authorized under those plans, agreements, and licenses shall be designed to result in an overall net gain in the recovery of Hawaii's threatened and endangered species." Federal issuance criteria does not require net gain in recovery, however requires that the taking will not appreciable reduce the likelihood of survival and recovery of the species in the wild (PL 93-205, Section 10(a)(2)(B)).

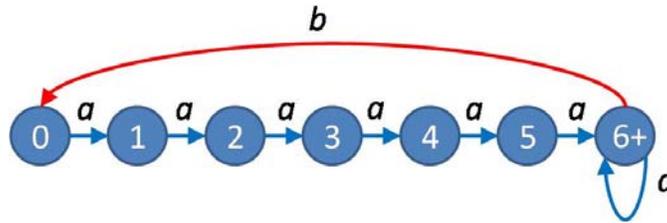


Figure 2-1 Stage-based model for Newell's shearwaters. Numbers refer to stage (0 through 5 represent sub-adult ages, 6+ adults), a) is survivorship S and b) fecundity.

$$N_t \begin{bmatrix} 0 & 0 & 0 & 0 & 0 & 0 & F_a \\ S_0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & S_1 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & S_2 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & S_3 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & S_4 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & S_5 & S_a \end{bmatrix}$$

Figure 2-2 Newell's shearwater population matrix used for model simulations. N_t is the stage-specific population size at time t , S is the age specific survival at ages 0, 1, 2, 3, 4, 5 and adult (6+), and F is fecundity, the probability of producing a viable one-year old in the next generation and defined as $F = f \times b \times c$, where f is the proportion of the population that is female (assumed to be 0.5), b is breeding probability, and c is the proportion of chicks that fledged from the eggs laid by a female parent.

Models were deterministic and made no allowance for natural variation in vital rates over time (i.e. stochastic models). Reproductive success is the most well documented aspect of petrel and shearwater biology that demonstrates this natural variation, typically showing wide variation from year to year (Appendix 1) because of fluctuations in oceanographic conditions affecting food supply (Warham 1990). A practical outcome of this variation is that it may make a small population more susceptible to extinction (i.e. several 'bad' years in a row). For our modeling purposes, we deliberately chose life history values that represented long-term means, and thus incorporated an element of long-term variation into models.

2.1.1 Stable, Growth and Optimal models

We developed a stable model based on estimated life history values for a population with no negative or positive growth. We also developed a growth and optimal breeding model to estimate a biologically reasonable and maximum growth rate, the latter under ideal conditions (i.e. no marine or terrestrial threats and a recovering population). Note that we did not consider this optimal growth rate achievable under any of our management scenarios, and the purpose was to demonstrate the limited opportunity

for growth in this species even under optimal breeding conditions, which is similar for other Procellariiformes (Bonnaud et al. 2009, Cuthbert and Davis 2002b, Keitt et al. 2002).

Stable Model:	$\lambda = \text{Life History}_{stable}$
Realistic Growth Model:	$\lambda = \text{Life History}_{growth}$
Optimal Growth Model:	$\lambda = \text{Life History}_{optimal}$

2.1.1.1 Elasticity and sensitivity analyses

To determine which vital rates are most significant to the population growth rate we conducted elasticity and sensitivity analyses using the stable model. Elasticity and sensitivity analyses are common modeling tools to assess which vital rate has the greatest impact on growth rate. For managers, these analyses are important tools to identify what vital rate to target with various management actions, and achieve the most impact for recovery (Keedwell 2004). For example, is it more important to reduce adult mortality from powerlines or fledgling mortality from light attraction?

2.1.2 Threat Models

To investigate the effects of powerline strike, fallout, and predation at the colony (independently, and in combination) on population growth rate, we adjusted the stable model for these three threats under four levels: none, low, medium, and high (threat models). For comparison, we provide Ainley's measures of these three threats (Ainley et al. 2001) recalculated in a Lefkovitch model in Appendix 2.

$$\text{Threat Models:} \quad \lambda = \text{Life History}_{stable} - \text{Threats}$$

To provide a comparison to population modeling results of threats, we compared growth rates produced in threat models to trends from the Save Our Shearwaters program and ornithological radar data collected on Kauai since 1993 (Appendix 3). Radar data and SOS were modeled over time (Year) and the parameter estimate. The 95% Confidence Intervals for Year provided an annual measure of population change, as a comparative measure to assess growth rates (λ) produced in the threat models.

2.1.3 Management Models

We investigated the potential impact of management actions including minimization of fallout (light abatement) and powerline strike, the SOS program to recover and re-release fallout fledglings, plus colony management actions of predator control and eradication, and chick translocation.

We considered each of these management models independently, and in combination. We expressed the potential impact as a change in growth rate (λ), and λ required to reach a stable growth rate. For a subset of combined management scenarios we expressed potential effect on those proportions of the population being managed. The effect was expressed as the difference in population size compared to no management actions (i.e. baseline). We used a baseline population experiencing high rates of powerline collision, fallout, and predation as this growth best fit annual population change suggested by ornithological radar and SOS trend modeling (Section 3.2.1 and Appendix 3). Further, because these represented our most conservative estimates of current threats to Newell's shearwaters, this effectively created a worst case scenario from which to plan management.

Efforts to reduce artificial lights have increased in the last five years (Erichsen and Griesemer 2009), but efforts to reduce powerline collision on Kauai have been minimal. However, the effectiveness of both types of minimization efforts have not been quantified. For our modeling we considered the none, low and medium values produced in the threat models representative of minimized powerline strike and fallout mortality.

For the SOS program and Colony Management Actions (CMA), we made Management Models by adjusting threat models. To estimate the effects of terrestrial threats and the benefits of SOS and CMA on population growth rate, lambda or population growth rate should ideally be measured in situations where threats and removal of threats are occurring independently and then compared (Natividad Hodges and Nagata 2001, Pierce 2002, Rayner et al. 2007, Zino et al. 2008). This opportunity does not exist for Newell's shearwaters because few CMA have been offered to any Newell's shearwater colony, and the effectiveness of the SOS program has not been quantified. In this study, colony management models were created by adjusting stable lambda values for estimated effects of terrestrial threats; and benefits of, whereby:

$$\text{SOS Management Model (Stable):} \quad \lambda = \text{Life History}_{\text{stable}} - \text{Threats} + \text{Management}_{\text{SOS}}$$

$$\text{Colony Management Model (Stable):} \quad \lambda = \text{Life History}_{\text{stable}} - \text{Threats} + \text{Management}_{\text{CMA}}$$

Theoretically under this modeling approach, the most beneficial management outcome would remove all the threats and allow the population to return to a stable growth rate (i.e. the maximum growth achievable would be $\lambda = 1.000$). For example, this approach would demand that even if 100% of Newell's shearwater predation could be alleviated on Kauai, should there be any mortality from fallout and powerline collision, the population would still decline (albeit slower).

To identify what growth rates would be required to stop a decline under our threat models, we developed a second set of colony management models (and not SOS models) where fecundity and survivorship values were increased (*Life History_{growth}*) until a stable population growth rate was achieved. We then compared this to our Realistic and Optimal Growth model to provide context for what is biologically practical.

$$\text{Colony Management Model (Required Growth): } \lambda_{\text{Stable}} = \text{Life History}_{\text{growth}} - \text{Threats} + \text{Management}_{\text{CMA}}$$

We also created chick translocation models whereby chicks are moved to a site free of terrestrial threats (no predators, lights or powerlines) and subsequently return to breed there (Miskelly et al. 2009). We allowed these populations the life history values of our Realistic Growth Model because they were subject to no terrestrial threats.

$$\text{Translocation Management Model (Growth):} \quad \lambda = \text{Life History}_{\text{Growth}}$$

2.2 Modeling assumptions

Assumptions and limitations specific to model inputs are described in each relevant report section. Other key assumptions include use of a closed population and a stable age structure at year one. Input limitations include no estimation of adjustments from habitat modification or marine threats.

Our modeling approach required assumption of a closed population because it was not possible to estimate immigration or emigration from other islands. However we expect immigration and emigration would have little impact on the overall modeling outcomes because of the high site fidelity common to Procellariiformes, that 90% of the entire population breeds on Kauai, and that the remaining 10% breeding on other Hawaiian Islands are also likely subject to the same threats (Mitchell et al. 2005).

We also assumed that the population age structure at year one was stable because no estimate of current population demography is possible. Because the population is in decline and the threats facing birds affect different age classes disproportionately, this assumption is likely untrue. For example, if powerline mortality affects a greater number of sub-adults aged 2-5, then we would expect to see lower representation of these age classes in the population.

We could not estimate at-sea threats because of lack of information. Newell's shearwaters spend up to 80% of their life at-sea. This species is considered a yellow-fin tuna (*Thunnus albacores*) associate and a flock-feeder in the Eastern Tropical Pacific (ETP) (Birdlife International 2010a, Spear et al. 2007). Impacts from fisheries are unknown but are hypothesized to include reduced density of prey aggregations available to Newell's shearwaters (Spear and Ainley 2007). Such a factor would increase required prey searching and foraging effort to provision offspring, ultimately affecting annual reproductive success and adult survival. Newell's shearwaters are not as efficient a flyer as other 'tuna birds' (Spear and Ainley 1997), b) and, therefore, increasing prey search is especially perilous to them. This is a critical assumption to our modeling efforts because fisheries impacts may have been occurring since the advent of industrialized fishing in the Pacific in the 1950's. No population measures are available near this time period, although measures of genetic diversity and stable isotopes (dC13 and dN15) should offer valuable insights (Wiley et al. 2011). For our population modeling, any fisheries effects would operate as an additional unquantified threat, potentially leading to an overestimation of modeled threats of predation, powerline strike or fallout, or underestimation of total population growth rate. Investigating possible fisheries effects is important to recovery planning.

A second unquantified threat is climate change. Generalized effects from global climate change that could affect Newell's shearwaters include increased thermal stratification, lower ocean productivity (Sarmiento et al. 2004), and possibly increased frequency of El Niño conditions which would affect ocean productivity (Devney et al. 2009)). Determining the potential effects of climate change is critical to recovery planning for Newell's shearwaters.

Although habitat modification is considered a threat to forest breeding Hawaiian petrel (*Pterodroma sandwichensis*) and Newell's shearwaters, no estimation was made for habitat modification because of a lack of quantified estimates. When compared to active Newell's shearwater colonies, three inactive Newell's shearwater colonies had higher rates of non-native vegetation (Holmes et al. 2009). Certain non-native plant species pose a threat by altering vegetation and habitat structure, for example young, fast growing strawberry guava (*Psidium cattleianum*) thickets reduce the burrowing habitat available to Hawaiian petrels (Penniman 2010). Further, fruiting species such as strawberry guava have a synergistic interaction with non-native mammals (Nogueira-Filho et al. 2009).

Further, we made no allowance for effects of rare catastrophic effects, such as hurricanes, or effects of inbreeding depression for small populations.

2.3 Data Quality

The quality of the data and assumptions used in any population model are key to their accuracy and predictive power (Keedwell 2004, Morris et al. 1999). Obtaining adequate life history data, and other important modeling parameters, is a key consideration before undertaking any modeling (Coulson et al. 2001, Morris et al. 1999). Key model inputs for our process included life history, threat levels and colony management effectiveness (Table 2-1). Spatial and temporal limitations are discussed here with specific limitations identified in each relevant report section.

The minimum life history data requirements for undertaking simple deterministic population modeling include age or stage structure, age of first breeding, mean fecundity (i.e. breeding success, breeding probability) for each age or stage, and mean survival for each age and stage (Caswell 2001, Keedwell 2004). For Newell's shearwaters direct measures of fecundity exist from studies at the Kalaheo colony (Telfer 1986, Ainley et al. 2001), and more recently from three pairs of Newell's shearwaters at Kilauea Point NWR (Zaun 2007) (Appendix 4). Importantly, however, the Kalaheo colony measures were in the presence of predation, and this colony appears abandoned, suggesting these fecundity measures do not represent stable breeding conditions. To estimate a stable population, we supplemented Newell's shearwater metrics with vital rates from analogous species under stable conditions. Survival has not been estimated for Newell's shearwaters, and we used survival from analogous species as was done by Ainley et al. (2001). Using metrics from analogous species is a common approach for many modeling efforts with Procellariiformes (Bonnaud et al. 2009, Jones 2002, Keitt et al. 2002, Martinez-Gomez and Jacobsen 2004), and justifiable given that many other petrel species will share similar life history (Brooke 2004). To account for the inherent natural variation in measures of survival and fecundity we used studies from analogous species where more than three years were available.

For our modeling purposes, we also quantified threats of predation, fallout and powerline mortality. Important limitations exist in our threat estimations, including the representativeness of the effects of predation observed at the Kalaheo colony, the spatial variation in predation across the island, changes in light and powerline infrastructure since the Ainley et al. (2001) mortality estimates, the population decline and proportional representation of the population facing powerline and fallout threats on Kauai, and changes in the likelihood of collecting both dead and live Newell's shearwaters. No quantification exists of these potential limitations.

The previous estimates of powerline and fallout mortality produced by Ainley et al. (1995, 2001) are likely less representative of current mortality. Factors that have likely decreased the rate of mortality include increased use of shielded lights across Kauai (KIUC STHCP 2010), increased tree growth adjacent to powerlines following Hurricane Iniki, and the lowering of some powerlines closer to vegetation or other structures (KIUC STHCP 2010). Factors likely to have increased collision mortality include increased development on the island especially near Poipu, Princeville and Kapaa, upgrading >20 km of major transmission powerlines to >15.2 m in height along powerline road, and installation of new powerlines on the island (KIUC STHCP 2010).

The rate of dead birds reported to SOS by the public also may have changed since 1993. Ainley et al. (1995) noted that 7-50% of dead birds were reported by the general public. Since then, the general public may be less likely to report dead birds given that finding them has become much less frequent; in addition law enforcement personnel have increased their effort since 2004.

In 1993, Ainley et al. (1995) estimated that 80% of the Newell's shearwater population bred north of Kealia. Since 1993, colonies on the East and South Shores show signs of abandonment (Holmes et al. 2009). Thus, the proportion of the Kauai population on the East and South shores may now be less than 20%, in which case the effect of powerline and fallout mortality on the total Kauai population may be lower because of reduced shearwater traffic on these shores.

Table 2-1 Data quality for model inputs

		Direct		Indirect
	Measure	Recent (>2000)	Historic (<2000)	Estimated and analogous measures (report section)
Baseline Life History	Adult Survival			Section 2.4.2.5 and Appendix 5
	Juvenile/Subadult Survival			Section 2.4.2.4
	Breeding Success	Kilauea Point 97-09	Kalaheo colony 80-84, 93-94	Section 2.4.2.3 and Appendix 1
	Breeding Probability	Kilauea Point 97-09	Kalaheo colony 80-84, 93-94	Section 2.4.2.2
	Age of first breeding			Section 2.4.2.1
	Population Size		At-sea studies 84-93	Section 2.4.1
Threat Rates	Powerline		EPRI studies 93-94	Section 2.4.3.3
	Fallout	SOS Database 1979-2009	SOS Database 1979-2009, EPRI studies 93-94	Section 2.4.3.2
	Predation		Kalaheo colony 80-84, 93-94	Section 2.4.3.1
Colony Management Effectiveness	Adult Survival			Section 2.4.2.5 and Appendix 5
	Juvenile/Subadult Survival			Section 2.4.2.4
	Breeding Success	Kilauea Point 97-09	Kalaheo colony 80-84, 93-94	Section 2.4.2.3 and Appendix 1
	Breeding Probability	Kilauea Point 97-09	Kalaheo colony 80-84, 93-94	Section 2.4.2.2

2.4 Model Inputs

2.4.1 Population Size

The only Newell's shearwater population estimate available is 84,000 individuals (19,300 breeding pairs in the spring and 16,700 breeding pairs in the autumn) based on at-sea observations between 1984 and 1993 (Spear et al. 1995). Since these at-sea estimates, population indices (e.g. ornithological radar, SOS fledglings) have declined by 75% from 1993-2009 (Day et al. 2003, Holmes et al. 2009). Recently, Pyle and Pyle (2009) suggested the breeding population of Newell's shearwaters on Kauai at 10,000 pairs, however this would suggest a decline of 40-48% during the past 15 year and is inconsistent with declines evident in indices of SOS and ornithological radar (Section 3.2.1 and Appendix 3).

For our modeling purposes, we adjusted initial population size to 25% of the Spear et al. (1995) estimate for Kauai, and assumed that 10% of the population bred elsewhere, producing 18,900 individuals, or 12,040 birds of breeding age under a stable age distribution (Ainley et al. 2001) (Section 2.2).

In addition to the limitations in using radar and SOS data to adjust population size outlined above, an additional limitation is spatial coverage. SOS and radar population trend measurements are produced from information from the southern, eastern and northern shores of Kauai, with limited to no coverage along the Northwestern shore where significant numbers of Newell's shearwaters breed (KESRP, unpublished data). While the colonies on the northwestern shore are subject to predation and light attraction, powerline collisions are minimal because of the lack of infrastructure on Kauai's northwestern shore. Should passage rates on the Northwestern shore not follow these trends, we would expect that our adjusted population size may be an underestimate. Repeating at-sea surveys or determining another method of population estimation is critical to recovery planning.

2.4.2 Life History Parameters

2.4.2.1 Age of first breeding

Age of first breeding within Procellariiformes species can vary between sites and between years (Warham 1996). Age of first breeding also can be a density dependent factor in Manx shearwaters (*Puffinus puffinus*), a closely related species, as a result of burrow competition (Brooke 1990) and in wandering albatrosses (*Diomedea exulans*) because of fishery related mortality (Croxall et al. 1990, Weimerskirch and Jouventin 1987).

We assumed the age of first breeding to be 6 years, similar to Ainley et al. (2001) and comparable vital rates from Manx shearwaters (Brooke 1990), and short-tailed shearwaters (*Puffinus tenuirostris*) (Bradley et al. 1989).

2.4.2.2 Breeding probability

Breeding probability is the probability that any one bird will breed in a given year (Newton 1998). This is inherently difficult to measure in burrowing petrels because it relies on documenting the *absence* of a breeding event, and can be confounded by mortality, failed breeding, sub-adult burrow occupancy and Type II error (failure to observe a breeding event when there was one). Further, breeding probability is dynamic and in seabirds can be influenced by extrinsic events such as climatic conditions (e.g. El Niño

decreasing forage availability and breeding likelihood (Ainley and Boekelheide 1990), or changes in habitat availability (e.g. hurricanes damaging habitat reducing burrowing opportunities (Ainley et al. 1995)). Demographic studies of long-lived seabirds have shown that measures of reproductive performance, including breeding probability, and survivorship are a function of individual quality, age and experience. For example, the likelihood of breeding increases with age (Ainley and DeMaster 1980), and less successful breeders may breed less often (Lescroel et al. 2009).

Accurate measures of breeding probability for long-lived seabirds typically come from long-term studies and are notably sparse. Amongst studies of shearwaters, breeding probability has been reported at ~86% (n>40 years) for short-tailed shearwaters (Bradley et al. 2000), 86-90% (n=15 years) in Cory's shearwater (*Calonectris diomedea*) (Mougin et al. 1997), and ~80% (n=~14 years) for the Manx shearwater (Brooke 1990). Amongst 15 estimates of breeding probability in Procellariiformes, a mean of 82% was calculated (SD=8%, Table 2-2).

Table 2-2 Breeding Probability in Procellariiformes.

Species	Breeding Probability	Reference
<i>Diomedea exulans</i>	87%	Jouventin & Weimerskirch (1988) ¹
<i>Diomedea chlororhynchos</i>	90%	Jouventin & Weimerskirch (1988) ¹
<i>Diomedea melanophrys</i>	80%	Prince et al. (1994) ¹
<i>Diomedea chrysostoma</i>	72%	Prince et al. (1994) ¹
<i>Diomedea immutabilis</i>	81%	Fisher (1969) ¹
<i>Phoebastria palpebrata</i>	75%	Jouventin & Weimerskirch (1988) ¹
<i>Phoebastria fusca</i>	83%	Jouventin & Weimerskirch (1988) ¹
<i>Fulmarus glacialis</i>	95%	Hatch (1987) ¹
<i>Pterodroma sandwichensis</i>	89%	Simons (1984)
<i>Calonectris diomedea</i>	90%	Mougin et al. (1997)
<i>Puffinus tenuirostris</i>	86%	Bradley et al. (1990)
<i>Puffinus puffinus</i>	80%	Brooke (1990)
<i>Puffinus mauretanicus</i>	74%	Oro et al. (2004) ²
<i>Bulweria bulwerii</i>	83%	Jouventin & Weimerskirch (1988) ¹
<i>Hydrobates pelagicus</i>	69%	Hemery et al. (1986) ¹

1. In Mougin et al. (1997), reported as proportion of sabbatical birds.
2. Reported as skipping rate.

At Kilauea Point National Wildlife Refuge, three pairs of Newell's shearwaters are currently known to breed. Pairs were first identified in 1997, 2001 and 2008, and since that time have bred for 12, 9 and 2 years, respectively, producing a breeding probability of 100% (Appendix 4, Table 3). This value, plus their outstanding reproductive success, is likely not representative of the entire breeding population, nor as a realistic management target. Outcomes of 100% reproductive success and 100% breeding probability were not identified in any of the 52 studies we found for burrowing petrels (Table 2-2 and Appendix 1). These Kilauea birds may be 'super-breeders' (Lescroel et al. 2010, Lescroel et al. 2009): members of the population that will consistently have higher reproductive output based on inherent individual quality.

Few direct measures of breeding probability exist for Newell's shearwaters. Telfer's (1986) study consisted of checking burrows previously known to have hosted breeding in 1980 or earlier, and he reported that from 1981-1985, an average of 46.6% of these burrows were active (Appendix 4, Table 1). Ainley et al. (2001) re-visited Telfer's (1986) data, and assumed that not all active burrows would have held breeders, and adjusted this figure based on 15.9% of the population being 4 and 5 year olds that would have occupied burrows but not bred, to produce a breeding probability of 54.7%.

Breeding probability of 46.6 or 54.7% for Newell's shearwaters is low in comparison with values reported for other Procellariiformes and *Puffinus* species reported in Table 2-2. When 54.7% was used by Ainley et al. (2001) in their modeling efforts, it was considered a key factor limiting population growth (a value of 80% from the Manx shearwater (Brooke 1990) was needed to simulate a stable population). Both Ainley et al. (2001) and Telfer (1986) identified mate loss from the high adult predation rates reported at the colony as contributing to this outcome. Mortality caused by nearby powerlines is also likely to have contributed to this low rate. Ainley et al (1995) reported 10 dead sub-adults and adults killed per km from nearby Lawai in the same flyway as Kalaheo colony.

We followed Ainley et al. (2001) in using a breeding probability of 80% from the closely related Manx shearwater (Brooke 1990) to model a stable population. We also considered 80% a biologically reasonable management target (i.e. breeding under favorable conditions), and made reductions in breeding probability for areas not subject to predator control correspondingly (Section 2.4.4.4 and 2.4.4.5).

2.4.2.3 Reproductive success

Reproductive success is the number of chicks fledged from eggs laid, and can vary widely among colonies and years (Warham 1996) because of food availability (Newton 1989) and intrinsic factors including increased performance with age, individual quality and experience (Wooller et al. 1989).

Brooke (2004) states that values between 40–70% are typical of shearwaters and petrels, with no consistent difference in this metric among the four families of Procellariiformes. Mean reproductive success of burrowing Procellariiformes for studies ≥ 3 breeding seasons in areas where predators never occurred was $0.59 \pm 0.11SD$ ($n=17$) and $0.62 \pm 0.08SD$ ($n=9$) for areas where predators were eradicated (Table 2-5 and Appendix 1). The long term study (≥ 40 years) of short-tailed shearwaters on Fisher Island, Australia, provides average breeding success at a colony in the absence of predators as well as inter-annual variation. Wooller et al. (1989) reported a mean of 61% (range: 25-83%), which is similar to the mean value of the above studies.

Reproductive success of the Newell's shearwaters at the Kalaheo colony ranged from 54 – 59% (Ainley et al. 1995, Telfer 1986) (Appendix 4, Table 1 and 2). Reproductive success of three pairs of NESH at the KPNWR was 100% for 23 breeding attempts (Zaun 2007) (Section 2.4.4.4 and Appendix 4, Table 3). Similar to breeding probability, we considered 100% breeding success unrepresentative of the entire breeding population, and an impractical target to consider for colony management actions. In the closely related Manx shearwater, Brooke (1990) reported a mean reproductive success of 70.1% at Skolkholm Island and 53.9% at Rhum Islands.

To model a stable population, we used 60% breeding success, close to the mean from Table 2-5 of studies in areas where predators were never present, eradicated, or controlled. To model an optimal population, we used 70% breeding success and we considered this to represent the upper end of what is likely for the species when averaged over a minimum of 3 years.

2.4.2.4 Sub-adult survivorship

Sub-adults are those younger than breeding age. At breeding colonies, sub-adults increase attendance as they age, investing more time to gain breeding experience with each year. Away from the colony these birds must gain experience in foraging (Warham 1990, 1996). Developing these skills comes at a cost, and sub-adult survival in the first few years is typically lower than adult survival. Sub-adult survival is a difficult variable to measure because individuals are not reliably associated with specific burrows.

Brooke (1990) estimated that 33.3% of Manx shearwaters survived from fledgling to breeding age (age 6 or later). We used age-specific survival rates estimated in Ainley et al. (2001) to achieve Brooke's overall survival value. Ainley et al. (2001) maintained a pattern of age specific survival rates of 0.654, 0.78, 0.89, and 0.905 (years four, five and six), which we calculated at 0.337 survival from fledgling to breeding age.

2.4.2.5 Adult survival

Adult survivorship (S_x) is the proportion of adults in the population that survive to the following year. Survivorship in petrel species increases with adult body mass (Warham 1996). For Newell's shearwaters, Ainley et al. (2001) used adult *P. puffinus* S_x of 0.905 based on Brooke (1990). This value was also consistent with an estimated survivorship of Newell's shearwaters based on the allometric regression of 16 petrel species (with 5 species <1000 g) based on S_x data from Gaillard et al. (1989) and mass from Dunning (1992) (Ainley et al. 2001).

We consider that adult S_x of 0.905 for Newell's shearwaters in the absence of threats to be low. Indeed, with a reproductive success of 0.6 and a breeding probability of 0.8, this produces a population declining at 1% per year. A notable comparison is adult S_x in Hutton's shearwaters was measured at 0.931 (n=9 years) (Cuthbert and Davis 2002a). Previous estimates (as opposed to direct measures) used in other modeling efforts of adult S_x in 300-600 g shearwaters and petrels range from 0.92 – 0.94 (excluding the Balearic shearwater (*Puffinus mauretanicus*) facing significant threats of fisheries bycatch adult mortality) (Table 2-3).

Table 2-3 Previous estimates of Adult S_x in small shearwaters and petrels used in modeling

Species	Mass (g)	S_x	References
<i>Pterodroma atrata</i>	226	0.93	Brooke et. al (2010)
<i>Puffinus auricularis</i>	323	0.92	Kiett et. al (2002)
<i>Puffinus newelli</i>	381	0.905	Ainley et. al (2001)
<i>Puffinus yelkouan</i>	408	0.93	Bonnaud et al. (2009)
<i>Puffinus mauretanicus</i>	497	0.78	Oro et. al (2004)
<i>Puffinus carnipes</i>	609	0.94	Priddel et. al (2006)

We updated the allometric regression to include 41 data points (Appendix 5), including 23 measures for petrels <1000 g, plus updated mass measurements (Dunning 2008). Initial data screening highlighted one outlier (*Pelecanoides urinatrix*, S_x 0.67) which we removed. Adult S_x showed a significant positive relationship with $\ln(\text{mass})$ ($F_{1,38}=7.55$, $P=0.009$), however adjusted R^2 was only 0.144, highlighting that a large proportion of variation in adult S_x was not explained by $\ln(\text{mass})$ in our equation.

Based on this equation, the predicted adult S_x of Newell's shearwaters (381 g) is 0.920, with 95% CI for the mean at 0.909 and 0.931 (Figure 2-3). In combination with breeding success of 0.6 and breeding probability of 0.8 this produced a population just above stable (growth rate of 0.05% per year).

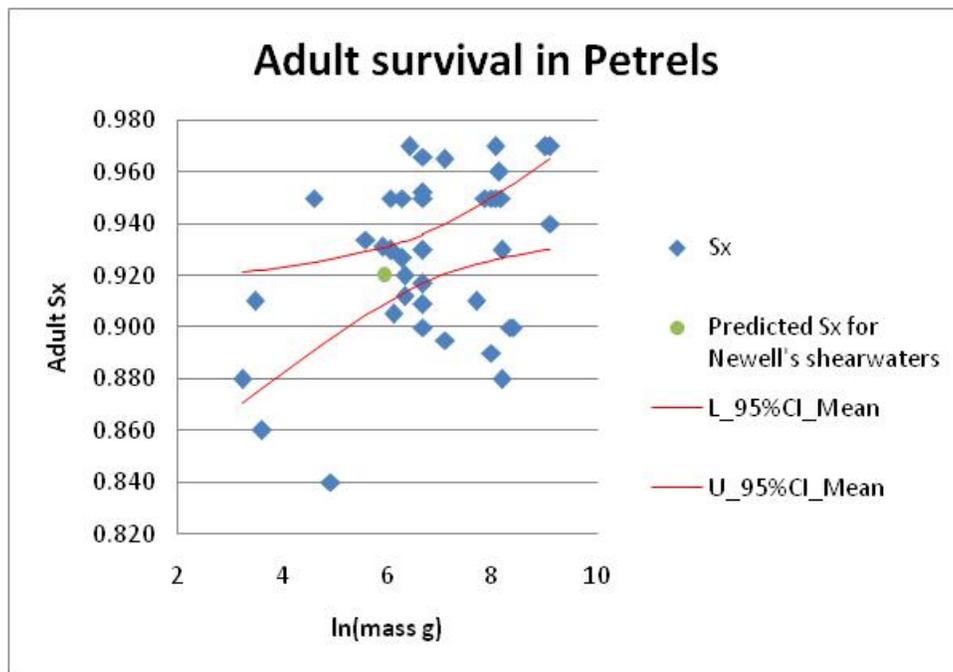


Figure 2-3 Relationship between adult survival and $\ln(\text{mass})$. $S_x = 0.00882\ln(\text{mass}) + 0.86754$

2.4.3 Threats

Our adjustments for predation, light and powerline mortality for Newell's shearwaters were primarily based upon Ainley et al. (2001). Adjustments from stable for threat models are given in Table 2-4.

Table 2-4 Adjustments from stable for threat models

		Breeding Probability	Reproductive Success	Survivorship						
				0 - 1	1 - 2	2 - 3	Age 3 - 4	4 - 5	5 - 6	6+ (adult)
Stable		0.800	0.600	0.6540	0.7800	0.8900	0.9050	0.9050	0.9050	0.9200
Predation	low	-0.100	-0.100			-0.0250	-0.0250	-0.0500	-0.0500	-0.0050
	med	-0.200	-0.250			-0.0500	-0.0500	-0.1000	-0.1000	-0.0100
	high	-0.300	-0.400			-0.0750	-0.0750	-0.1500	-0.1500	-0.0150
Powerline	low					-0.0060	-0.0060	-0.0060	-0.0060	-0.0005
	med					-0.0120	-0.0120	-0.0120	-0.0120	-0.0009
	high					-0.0172	-0.0172	-0.0172	-0.0172	-0.0013
Predation										
Low fallout	none		-0.068							
	low		-0.078							
	med		-0.090							
	high		-0.109							
Med fallout	none		-0.087							
	low		-0.100							
	med		-0.116							
	high		-0.140							
High fallout	none		-0.107							
	low		-0.122							
	med		-0.142							
	high		-0.171							

2.4.3.1 Predation

Predation by introduced mammals is a key threat to colonial breeding seabirds (Birdlife International 2010a). Cats and rats are consistently identified as major contributors to the decline of seabirds around the globe (Keitt et al. 2002, Natividad Hodges and Nagata 2001). On Kauai, evidence of cat and owl predation on Newell's shearwaters comes from almost every colony on Kauai, including the most remote sites (Holmes et al. 2009, Telfer 1986). To account for predation we adjusted breeding probability, breeding success and survivorship.

Under natural conditions, divorce leads to a lower breeding probability in the following year because of the time required to acquire a new mate (Brooke 1990, Warham 1996). Predation can have the same effect on a pair by requiring the remaining individual to also search for a new mate. Given the difficulty of obtaining breeding probabilities even under the most optimal conditions, it is not surprising that a reduction of breeding probability due to predation is rarely reported. However, given the low breeding probability value obtained at the Kalaheo colony, and that this was likely an artifact of the predation recorded there, we considered this an important adjustment. We reduced 0.8 breeding probability by 0.10, 0.20 and 0.30 reduction in breeding probability as a response to predation, with our medium to high measure providing a comparative measure to 54.7% calculated by Ainley et al. (2001).

Chick and egg mortality from non-native predators is commonly reported in burrowing petrels (Warham 1996). We conducted a literature review to determine reproductive success in the presence and absence of predators (primarily cats, rats and mice) and after predator control and eradication (n=72, Appendix 1). From this literature review, for studies ≥ 3 years in duration, the mean breeding success of burrowing Procellariiformes in the presence of predation without predator control or eradication was 32% (n=17 studies, Table 2-5). While this broad scale effort did not isolate the effects of non-native predation from other threats that can influence breeding success (e.g., trampling of burrows by feral donkeys or overgrazing by rabbits leading to erosion and burrow collapse), predation was a key factor resulting in reduced reproductive success. While these predation impacts, and outcomes from colony management actions, will have limited direct transfer to Newell's shearwaters because of differences in predator species composition, habitat type, seabird species composition and other factors, we considered these results relevant as a guide to the scale of impact that predation can have on colonial breeding Procellariiformes weighing <1000 g, and for our modeling efforts.

Table 2-5 Breeding success for 44 studies of burrowing petrel ≥ 3 years. Individuals studies detailed in Appendix 1

	X	SD	N	Lower1SD	Upper1SD
Predators never present	0.59	0.11	9	0.48	0.70
Predators present	0.32	0.17	17	0.15	0.49
Predators eradicated	0.62	0.08	9	0.54	0.70
Predators controlled	0.58	0.13	9	0.45	0.71

Chick mortality has been recorded for Newell's shearwaters at forest colonies on Kauai; including chicks killed in burrows (KESRP, unpublished data). At the Newell's shearwater Kalaheo colony, predation was primarily associated with sub-adult and adult birds and not with chicks. Breeding success at Kalaheo in the presence of predation was 59% (n=5, 1981-1985) and 54% (n=2, 1993-1994), not much lower than 60% considered for a stable population, and suggesting little predation on chicks during these studies. However predation of burrowing petrel eggs and chicks is commonly recorded elsewhere (Appendix 1) and reasonable to expect for Newell's shearwaters given this age class has no means of escape from predators once located, and are naïve to this kind of threat. We adjusted breeding success by -0.10, -0.25 and -0.40 from stable in response to predation, similar to one SD above and below the mean from studies in Table 2-5.

Adult and sub-adult mortality from predation at the colony will vary based on phenology and age class (Bonnaud et al. 2009, Cuthbert and Davis 2002b, Martinez-Gomez and Jacobsen 2004). Despite less time spent at the colony, sub-adults are considered at higher risk from predation because they are more vocal. Adults are predicted to be most vulnerable during arrival and prospecting when they vocalize to re-establish pair bonds. Differential predation was estimated by Ainley et al. (1995) based on the number of dead sub-adults and adults found compared to the estimated number of birds at the colony (600). He reported an estimated predation rate of 2 and 3 year olds at 5%, for 4 and 5 year olds at 10%, and breeders at 1%. We used these rates, and adjusted 50% above and below these values to simulate higher and lower predation (-7.5%, -15%, and -1.5%; and -2.5%, -5%, and -0.05% by respective age classes).

2.4.3.2 *Fallout / Light attraction*

Fledgling fallout mortality was previously estimated by Ainley et al. (2001) using SOS numbers, and was considered to be a function of both discovery and morbidity, whereby discovery is the proportion of total fallout fledglings found and morbidity is the proportion of the total dead fledglings reported to SOS. Based on those estimates of mortality, Ainley et al. (2001) decreased reproductive success to model the effect of fallout, and we follow that method here. Because we considered predation to decrease reproductive success (i.e. increase chick mortality), we calculated fallout mortality estimates for each predation level (none, low, medium, high).

We considered four levels of discovery, where 100%, 80%, 67%, and 50% of all downed fledglings are reported by the SOS program, following Ainley et al. (2001). The current 5 year average (2005-2009) of the number of birds reported to SOS was 280, leading to 280, 350, 418, and 560 fallout birds at these discovery rates, respectively. Total island chick production under our no, low, medium and high predation models equaled 2889, 2107, 1264 and 602 fledglings produced respectively, and Table 2-6 shows the total fledgling fallout as a proportion of the total fledgling production on Kauai under these scenarios.

Table 2-6 Fallout as a proportion of total annual Newell's shearwater chick production, based on chick predation in the colony and discovery rate of fallout birds by the public, and a 75% population decline.

			Predation			
			None	low	medium	High
Discovery	Downed SOS fledglings	Fledglings in population	2889	2107	1264	602
1.00	280	Proportion of fledglings as fallout birds	0.10	0.13	0.22	0.47
0.80	350		0.12	0.17	0.28	0.58
0.67	418		0.14	0.20	0.33	0.69
0.50	560		0.19	0.27	0.44	0.93

Of the 16 estimates of the proportion of fallout birds of total fledgling production, we selected the 17th, 50th, and 83rd percentiles of the range of fallout (280-560) at each predation level to represent low, medium, and high fallout, similar to Ainley et al. (2001). For consistency in modeling threats as rate adjustments, we present the *decrease in reproductive success rates* at each level of fledgling population size rather than the *proportional decreases in reproductive success* (Table 2-7).

Table 2-7 Newell's shearwater reproductive success adjustments for fledgling fallout mortality used in threat and management models

	Fallout Threat level	No. of dead fledglings	Percentile of value range	Predation			
				None	Low	medium	High
Fallout	Low	327	0.17	-0.068	-0.078	-0.090	-0.109
	Medium	420	0.50	-0.087	-0.100	-0.116	-0.140
	High	513	0.83	-0.107	-0.122	-0.142	-0.171

2.4.3.3 Powerline strike

Ainley et al. (2001) previously estimated mortality from powerline collision and we retained those estimates here because no quantified search efforts have been undertaken since. These estimates were obtained based on standardized sampling (specific search routes under lines) and the dead birds collected. The numbers of dead birds found and reported were adjusted based on the probability of sighting a carcass, survey period and distance of powerlines not searched. The scaled adjustments resulted in low, medium, and high mortality rates based on the proportion of dead birds represented in the total Kauai sub-adult and adult population. Mortality estimates for subadults (ages 2 to 5) were 0.60%, 1.20%, and 1.72% and for adults were 0.046%, 0.092%, and 0.131% respectively.

2.4.4 Management actions as independent scenarios

Management actions considered were a) fallout and powerline strike minimization, b) the Save Our Shearwaters fallout fledgling rescue program, and c) colony based activities of predator eradication, predator control and translocation. We created a range of effectiveness scenarios for each management action.

2.4.4.1 *Fallout Minimization*

Fallout minimization is achieved by reducing the amount of upward artificial light available to Newell's shearwater fledglings susceptible to confusion by lights. This management action includes turning lights off, retrofitting lights with shielded fixtures, replacing lights with full cutoff, angling lights down, changing duration of light usage (timers), or changing light color. Previous studies of shielding at resort facilities have demonstrated that fallout can be reduced by nearly 40% (Reed et al. 1986). Fallout minimization actions can be accomplished through outreach and the HCP process. On Kauai, outreach to entities with the highest risk of light attraction is currently undertaken by KSHCP staff. Between 2005 and 2007 approximately 30 businesses on Kauai turned lights off and/or angled lights down to the ground (KSHCP 2010).

We considered the fallout levels of none, low or medium as representative of impact minimization given this is a preventative measure.

2.4.4.2 *Powerline Strike Minimization*

Powerline strike minimization is removing (undergrounding) lines, reconfiguring lines, or placing bird diverters on lines to eliminate or reduce collision risk. Undergrounding lines permanently eliminates risk of collision. Reconfiguring lines includes decreasing the line height down to the level of the surrounding vegetation, or lowering or switching from a vertical to horizontal grouping (Ainley et al. 1995). Bird diverter technologies have been demonstrated to have positive effects for some bird species by making lines more visible, thereby reducing collision risk (Yee 2008). Procellariiformes did not occur in the Yee (2008) study, and while bird diverters have been installed on some powerlines on Kauai, no quantification of their effectiveness for Newell's shearwaters has been measured.

The utility cooperative of Kauai has recently identified potential reconfiguration options for line segments (KIUC STHCP 2010) as previously suggested by Ainley et al. (1995). Implementation of these options is conditional on the utility cooperative's short-term HCP permit issuance (KIUC STHCP 2010).

Similar to fallout, we considered none, low and medium powerline strike representative of impact minimization given this is a preventative measure.

2.4.4.3 *Save Our Shearwaters program*

The Save Our Shearwaters program is a public conservation effort whereby fallout fledglings are recovered by members of the public, and then re-released. The program has been in operation since 1979, with more than 30,000 Newell's shearwater fledglings collected. No effort to quantify the effectiveness of the program has been undertaken.

Observations to determine the survival of fallout fledglings released under the SOS program are notably sparse. From 1979-2009, of 1094 adult and sub-adult (i.e. ≥ 1 year in age) mortalities collected, only 19 (4.14%) were banded in a previous year by SOS staff. Ainley et al. (1995) reported no previously banded SOS birds from 42 and 33 nests monitored at the Kalaheo colony in 1992-1993, despite 1000's of birds banded in that region for 13 years under the SOS program.

It is unlikely that fallout fledglings have equivalent survival to fledglings not experiencing fallout. It is reasonable to expect that many fallout fledglings released through the SOS program would have had a decreased survivorship because of undetected injuries, decreased health values (weight, hydration), or secondary complications acquired during fallout (disease, parasites, decreased waterproofing). Further, fledging is likely a critical 'learning' period in petrel life history, particularly given that birds spend the next two years at sea, and it is unclear if the fallout experience may impede future navigation or phototropic feeding abilities.

The assumption that all fallout fledglings released through the SOS program die before reaching breeding age (or 0% SOS effectiveness) over the entire course of the program from 1979-2009 is also likely untrue. More recently, SOS release protocols were modified to increase likelihood of survival, including feeding birds falling below minimum mass thresholds before they can be released (SOS 2010). Fledgling mass correlates highly with sub-adult survivorship in other seabirds and this may increase survival of these fledglings (Dann and Cullen 1990).

While we modeled 0% effectiveness for the SOS program in our combined management models (Section 2.4.3.2), we also modeled 100% to investigate the *theoretical* range of effects this program could offer. To model 100% effectiveness we reduced fledgling success (Table 2-8), based on the mortality of birds not being found. The mortality of the downed fledglings not found was estimated by combining the discovery levels (Section 2.4.3.2) with morbidity values which produced 64 mortality rates for each level of predation (Appendix 6). To represent low, medium, and high fallout, we selected the 17%, 50%, and 83% percentile of the range of mortality values at each predation level (Table 2-8). The theoretical management effect was the difference between 0% effectiveness and 100% effectiveness.

Table 2-8 Newell's shearwater reproductive success adjustments for fledgling fallout mortality used to investigate effect of SOS management action

	Fallout Threat level	No. of dead fledglings	Percentile of value range	Predation			
				none	Low	medium	High
Fallout with SOS program (100% effectiveness)	Low	58	0.17	-0.012	-0.014	-0.016	-0.019
	Medium	131	0.50	-0.027	-0.031	-0.036	-0.044
	High	204	0.83	-0.042	-0.048	-0.057	-0.068

2.4.4.4 Predator control

Predator control for Newell's shearwaters is the removal of cats, rats and barn owls from a colony site. This activity is only successful for the duration of application and is considered to have no value once stopped. Examples of predator control effectiveness are outlined in Appendix 1 and in Table 2-5.

Examples of predator control efforts for Newell's shearwaters range considerably between the Kilauea Point National Wildlife Refuge, Kalaheo colony and Upper Limahuli Preserve. At Kilauea Point National Wildlife Refuge, a fence keeps dogs at bay and predator control (rat bating, cat trapping and owl shooting) has ranged from none to fair since 1997 (fair being multiple traps set near burrows and checked daily throughout the year), with fair only applicable for a few years (Brenda Zaun, pers. comm.). Notably, even during the "fair" years, multiple photos of cats were obtained at the burrows highlighting luck and timing playing a role in these birds' continued survival. At Kalaheo colony, cat trapping was

implemented from 1982-1984 onwards following a significant predation event, and trap type, frequency, trap success and method are unknown, however likely consisted of cage traps checked during every monitoring visit (1-2 times per month, typically beginning in mid-late incubation). At the Kalaheo colony from 1993-94, predator control consisted of cage trapping also checked during regularly (every 2-3 days) during monitoring visits, with 1 cat caught in 1993 (Ainley et al. 1995).

At Upper Limahuli Preserve, predator control includes an ungulate exclusion fence, removal of noxious weeds (both initiated to protect watershed and flora habitat values), and cat, rat and owl control throughout the course of the breeding season. Predator control is limited by spatial and temporal coverage, and rodenticide application is not currently feasible under federal and state guidelines (NTBG 2006).

We considered predator control for Newell's shearwaters to adequately target each of these key predators in conjunction with habitat management (ungulate removal and noxious weed control). For our modeling purposes we considered 90% effectiveness as an upper limit. We expect that 100% of predation cannot be mitigated by predator control for Newell's shearwaters, given a) the dense forested colonies and steep terrain will make it unrealistic to obtain complete spatial coverage of cats, b) effective rat control is currently limited to bait stations thus limiting both spatial and temporal coverage, and c) fencing, trapping and baiting are ineffective for barn owls.

2.4.4.5 Predator eradication

Predator eradication at colony sites is the optimal management action, and has been achieved on offshore islets with positive effects for numerous seabird species (Clout and Russell 2006, Nogales et al. 2004). At 'mainland' seabird colonies, predator eradication has been achieved in combination with predator proof fences (Day and MacGibbon 2007) <http://www.xcluder.co.nz/>, with an important example for Hawaii completed at Kaena Point on Oahu in 2011. On Kauai, Newell's shearwater colonies are in steep and extremely rugged terrain, where predator-proof fences may be difficult to construct. To remove rats, significant regulatory hurdles must be overcome before aerial baiting can occur to remove rats once the fence is erected, and eradication via bait boxes will be problematic in many areas because of steep and rugged terrain. Predator-proof fences will not exclude barn owl predation on Newell's shearwaters and any such project would require investment in ongoing owl control. Despite these limitations, predator-proof fences yield great promise for recovery and HCP mitigation because they will a) protect against mongoose ingress should these predators become established on Kauai, b) likely provide a cost savings over the long-term when compared to perpetual predator control, and c) provide tremendous complementary benefits to other species affected by rats [e.g. Newcomb's snail (*Erinna newcombi*), *Pritchardia* palm species and native forest birds].

Large scale predator eradication is likely not achievable anytime soon, however small scale projects may be possible within the next 5-10 years. We considered the outcome from predator proof fencing (and subsequent ungulate and predator eradication) plus barn owl control 100% predator removal.

2.4.4.6 Chick translocation

Translocation of petrel chicks is a viable management action and recovery strategy for endangered petrels whereby a new population is established in a predator and threat free environment. Chicks moved ~one month before fledging (i.e. before they exit the burrow for the first time) will 'imprint' on the new location and return to the new site to breed. This has proven a successful strategy in Australia and New Zealand (Miskelly et al. 2009, Priddel and Carlile 2009) and has application for Newell's shearwaters.

A similar project was previously conducted with Newell's shearwaters eggs. Between 1979 and 1981, 90 eggs were moved to Kilauea Point National Wildlife Refuge, and placed under surrogate wedge-tailed shearwater (*Puffinus pacificus*) parents (Byrd et al. 1984). Today a minimum of three pairs are known to be breeding at the refuge (Zaun 2007) and are likely progeny of the cross-fostered birds. Observations suggest that additional birds are prospecting the area likely bolstered by the implementation of a social attraction project in 2007 to increase recruitment to this site (Haber et al. 2010).

Key benefits of translocation include expanding the range, establishing an 'insurance' colony and having a colony in an environment entirely free of terrestrial threats. Prospective sites for chick translocation include Lehua Islet (once rat free) and Kaena Point (following removal of predators). For our modeling purposes we investigated translocating up to 600 chicks over a 10 year period.

Key limitations to our translocation model were no allowance for natural recruitment, immigration, and no additional adjustment for vital rates. It is reasonable to expect the recruitment of prospecting (i.e. first time) breeders to the translocation colony may also occur independent of the translocated individuals (Miskelly et al. 2009, Priddel and Carlile 2009), and this will likely depend heavily on the location of the colony within an existing marine or terrestrial flyway. Should the translocated colony for example be Kilauea Point, we would expect this 'natural' recruitment to be higher than at Kaena because of the higher passage rates of Newell's shearwaters at the former site. Secondly, it is also reasonable to expect that the age of first breeding may decrease at a growing colony (Brooke 2010), because there are opportunities for younger birds to breed. Indeed, in July 2010 a three year old Newell's shearwater was recorded prospecting with a known breeder at Kilauea Point National Wildlife Refuge, providing the possibility it may breed at four years of age in 2011.

2.4.5 Management actions as combined scenarios

Each of the management actions modeled are not likely to be done in isolation, and will also be subject to practical limitations (i.e. predator eradication for the entire island is not feasible). We subsequently chose five models to identify some potential scenarios on Kauai, with two models expressing effect for the entire Kauai population (Table 2-9).

Models included a Northwest Kauai scenario, a predator proof fence, and a chick translocation project. The Northwest of Kauai is an obvious choice to implement management actions because of the high density of breeding observations in this region (Holmes et al. 2009), this area naturally abates for collision with artificial structures, and fledglings from this region may only be exposed to low artificial light coverage (Troy et al. 2011). The scale and topography of many areas of the Northwest of Kauai will

likely lead to only predator control being appropriate, and for this scenario we consider 0% SOS effectiveness. A predator proof fence for 1-10% of the existing population was modeled. With no powerline and no fallout threat given, the location of such a fence is unknown, but we anticipate it will be constructed in a region where maximum fallout and powerline minimization can also be achieved. Chick translocation was modeled for up to 60 chicks moved per year for 10 years to a predator, light and powerline free environment. Finally we considered two models for all three of these scenarios combined with 1) 10% population subjected to predator control, 5% to predator eradication, and 100 chicks translocated over five years, and 2) 20% population subjected to predator control, 10% to predator eradication, and 400 chicks translocated over 10 years.

Table 2-9 Combined management action scenarios modeled

Modeling Scenario	Colony Management Action	Fallout threat	Powerline threat	SOS effectiveness
Northwest PC	High predation, 90% predator control effectiveness, 1-20% of the population	low	none	0%
Predator Eradication (PE)	100% predator control, 1-10% of the population	none	none	n/a
Chick Translocation	Up to 600 chicks translocated over 10 years	none	none	n/a
All three combined				
Kauai 1	10% population predator control	low	none	0%
	5% population predator eradication	none	none	n/a
	100 chicks moved over 5 years	none	none	n/a
Kauai 2	20% population predator control	low	none	0%
	10% population predator eradication	none	none	n/a
	400 chicks moved over 10 years	none	none	n/a

3 Results

3.1 Stable realistic and optimal growth models

Simulations of stable and optimal models are depicted in Figure 3-1. The stable simulation represents an undisturbed population with a reproductive success of 60%, an annual breeding probability of 80%, and annual adult survival of 92%. The realistic growth model produced a growth rate of 1.2%, and the

optimal model 2.3% (Table 2-2, Table 2-3 and Table 2-5). Starting with a population of 1000 individuals, the realistic growth model produced 115 additional birds, and optimal growth model resulted in an additional 255 birds, after ten years compared to the stable model.

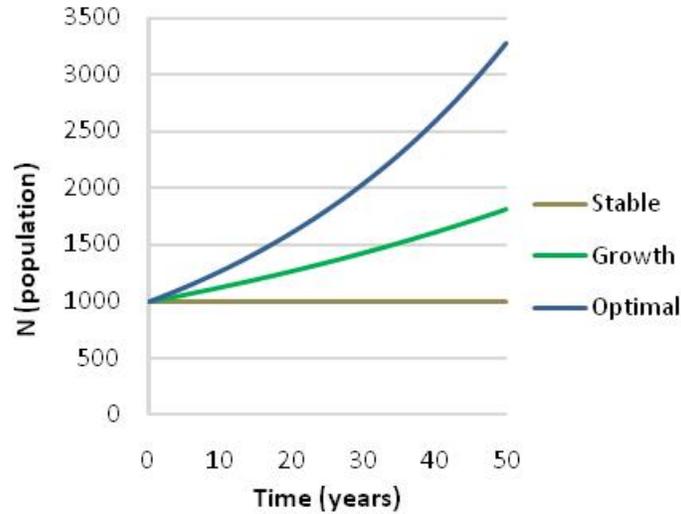


Figure 3-1 Simulation of stable, growth and optimal breeding models for Newell's shearwaters

3.1.1 Elasticity and sensitivity analyses

Lambda or growth rate was most sensitive to changes in adult survival in the stable model (Figure 3-2).

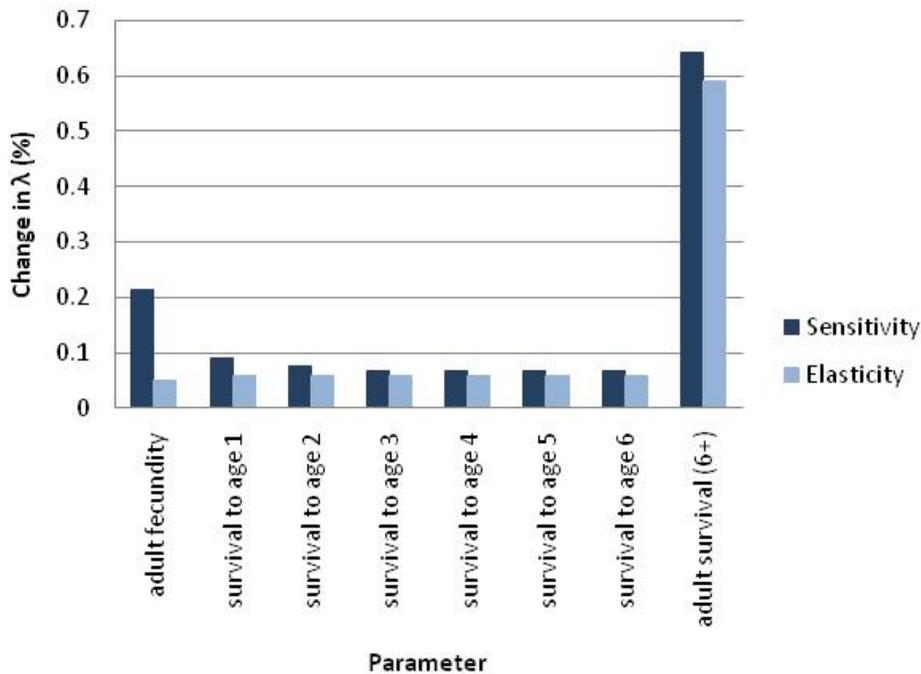


Figure 3-2 Elasticity and sensitivity of lambda to vital rates in stable population model

3.2 Threat models

Final lambdas for each threat model are presented in Table 3-1. Predation had the greatest impact on growth rate, with our low, medium and high predation rates producing declines of 2.7%, 5.5% and 7.9% per year (Figure 3-3). Fallout and powerline mortality produced declines of 0.6, 0.8 and 1% per year versus 0.1, 0.3, 0.4% per year, respectively. Ainley et al.'s (2001) recalculated threat models are presented in Appendix 2.

Table 3-1 Growth rates for Newell's shearwater under low, medium, and high projected levels of fallout, powerline and predation threats, as an adjustment from stable.

				Powerline			
				<i>None</i>	<i>Low</i>	<i>Medium</i>	<i>High</i>
Predation <i>None</i>	Fallout	<i>None</i>	<i>None</i>	1.001	0.999	0.997	0.996
		<i>Low</i>		0.994	0.993	0.991	0.989
		<i>Medium</i>		0.992	0.991	0.989	0.988
		<i>High</i>		0.990	0.989	0.987	0.986
Predation <i>Low</i>	Fallout	<i>None</i>	<i>None</i>	0.973	0.972	0.970	0.969
		<i>Low</i>		0.966	0.965	0.964	0.962
		<i>Medium</i>		0.964	0.963	0.962	0.960
		<i>High</i>		0.962	0.961	0.960	0.958
Predation <i>Medium</i>	Fallout	<i>None</i>	<i>None</i>	0.945	0.944	0.943	0.942
		<i>Low</i>		0.937	0.936	0.935	0.934
		<i>Medium</i>		0.935	0.934	0.933	0.932
		<i>High</i>		0.933	0.932	0.931	0.930
Predation <i>High</i>	Fallout	<i>None</i>	<i>None</i>	0.921	0.920	0.919	0.919
		<i>Low</i>		0.913	0.912	0.911	0.911
		<i>Medium</i>		0.910	0.910	0.909	0.909
		<i>High</i>		0.908	0.907	0.907	0.906

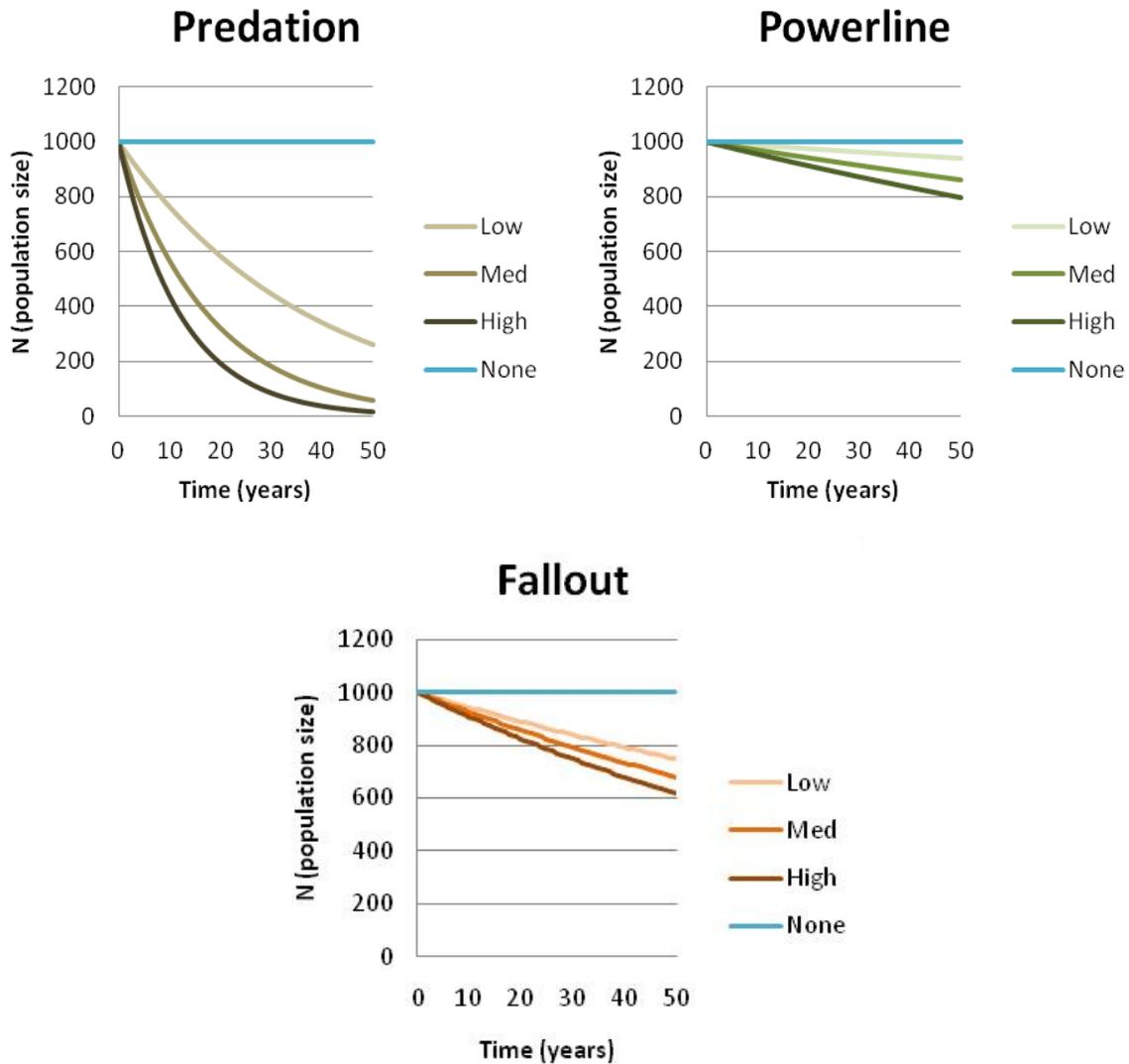


Figure 3-3 Effects of predation, powerline and fallout threats on growth rate, assuming a stable population.

3.2.1 Comparison to Ornithological Radar and SOS trends

Save Our Shearwaters and Ornithological Radar trend data suggest declines of 5.2 - 10.1% per year. When assessed independently or in combination, fallout and powerline threats alone (Table 3-1) produced far less conservative declines than suggested by SOS and radar trends (Table 3-2). Threat models using medium or high predation levels produced declines equivalent to Radar and SOS trends. The radar model using only data from the peak Newell's shearwater traffic period from 1993-2010 (Radar13.3&4.93-10) and the SOS model using data from 1988-2009 (SOS.88-99) (Table 3-2, Appendix 3) indicated declines equal to or worse (0.899, 0.905) than the lowest lambda produced in our threat models (0.906).

Table 3-2 Model terms and outcomes for fitting models of SOS and Ornithological Radar date by year¹.

Model	Years	Adjusted R ²	1 + Parameter estimated for the term year (λ)	Lower 95% CI	Upper 95% CI
Radar13.ALL. 93-10	1993-2010	.77	0.915	0.901	0.928
Radar13.3&4.93-10	1993-2010	.63	0.905	0.888	0.922
Radar13.ALL. 99-10	1999-2010	.75	0.948	0.927	0.968
Radar13.3&4.99-10	1999-2010	.62	0.923	0.898	0.947
SOS.79-09	1979-2009	.81	0.930	0.917	0.942
SOS.88-09	1988-2009	.91	0.899	0.885	0.914

1. Radar follows Day et al. (2001) methodology (all data, and data from suggested peak of Newell's shearwater traffic) for 1993-2010 and 1999-2010. SOS is the number of Newell's shearwater fallout fledglings recovered per year with no adjustment for discovery rate.

3.3 Management models

3.3.1 Management actions as independent scenarios

3.3.1.1 Fallout and powerline strike minimization

Minimizing fallout increased the growth rate by 0.4, 0.4 and 0.5% under none, low and high predation threats, respectively (Table 3-1). Minimizing powerline strike increased growth rates by 0.1% under high predation threats to 0.3% under low predation threats, and by 0.3% under no predation (Table 3-1).

3.3.1.2 Save Our Shearwaters program

The SOS program with 100% effectiveness has the potential to increase growth rate by 0.5% under no predation threats to 0.8% under high predation threats (Table 3-3).

Table 3-3 Effect of Save Our Shearwaters program on population growth rates under low, medium, and high projected levels of fallout, powerline, and predation threats

				Powerline			
				None	Low	Medium	High
Predation None	Fallout	Low		0.5%	0.5%	0.5%	0.5%
		Medium		0.6%	0.6%	0.6%	0.6%
		High		0.6%	0.6%	0.6%	0.6%
Predation Low	Fallout	Low		0.6%	0.6%	0.6%	0.5%
		Medium		0.6%	0.6%	0.6%	0.6%
		High		0.7%	0.7%	0.7%	0.7%
Predation Medium	Fallout	Low		0.6%	0.6%	0.6%	0.6%
		Medium		0.7%	0.7%	0.7%	0.7%
		High		0.8%	0.8%	0.7%	0.7%
Predation High	Fallout	Low		0.7%	0.7%	0.7%	0.6%
		Medium		0.8%	0.7%	0.7%	0.7%
		High		0.8%	0.8%	0.8%	0.8%

3.3.1.3 Colony management actions

In combination with powerline strike and fallout threat level, predator control produced population growth rates from 0.976 to 0.991. Predator eradication produced population growth rates from 0.986 to 1.00 (Table 3-4). Chick translocation produced a population growth rate of 1.012 because of the translocation to terrestrial threat free areas.

Table 3-4 Growth rates with chick translocation, predator eradication (100% effectiveness), and predator control (90% effectiveness) at low, medium, and high projected levels of fallout, powerline, and predation threats

				Powerline			
				None	Low	Medium	High
Chick Translocation	Predation <i>None</i>	Fallout <i>N/A</i>		1.012	n/a	n/a	n/a
Predator Eradication <i>100% Effectiveness</i>	Predation <i>All (Low, Medium, or High)</i>	Fallout <i>None</i>		1.001	0.999	0.997	0.996
		<i>Low</i>		0.994	0.993	0.991	0.989
		<i>Medium</i>		0.992	0.991	0.989	0.988
		<i>High</i>		0.990	0.989	0.987	0.986
Predator Control <i>90% Effectiveness</i>	Predation <i>Low</i>	Fallout <i>None</i>		0.998	0.996	0.994	0.993
		<i>Low</i>		0.991	0.990	0.988	0.987
		<i>Medium</i>		0.989	0.988	0.986	0.985
		<i>High</i>		0.987	0.986	0.984	0.983
	Predation <i>Medium</i>	Fallout <i>None</i>		0.994	0.993	0.991	0.990
		<i>Low</i>		0.988	0.986	0.983	0.983
		<i>Medium</i>		0.986	0.984	0.983	0.981
		<i>High</i>		0.984	0.983	0.981	0.980
	Predation <i>High</i>	Fallout <i>None</i>		0.991	0.990	0.988	0.986
		<i>Low</i>		0.985	0.983	0.981	0.980
		<i>Medium</i>		0.983	0.981	0.980	0.978
		<i>High</i>		0.981	0.979	0.978	0.976

3.3.2 Additional Growth required to achieve a stable population

Additional growth required to achieve a stable population under management scenarios in Table 3-4 are in Table 3-5.

Table 3-5 Growth rates required to achieve stable population in management areas under low, medium, and high projected levels of fallout, powerline, and predation threats. Results in italics are those equal to or greater than modeled realistic growth rate, and results in bold italics are greater than optimal growth rate.

				Powerline			
				<i>None</i>	<i>Low</i>	<i>Medium</i>	<i>High</i>
Predator Eradication <i>100% Effectiveness</i>	Predation All (<i>Low, Medium, or High</i>)	Fallout	<i>None</i>	0.0%	0.1%	0.3%	0.4%
			<i>Low</i>	0.6%	0.7%	0.9%	1.1%
			<i>Medium</i>	0.8%	0.9%	1.1%	1.2%
			<i>High</i>	1.0%	1.1%	1.3%	1.4%
Predator Control <i>90% Effectiveness</i>	Predation Low	Fallout	<i>None</i>	0.2%	0.4%	0.6%	0.7%
			<i>Low</i>	0.9%	1.0%	1.2%	1.3%
			<i>Medium</i>	1.1%	1.2%	1.4%	1.5%
			<i>High</i>	1.3%	1.4%	1.6%	1.7%
	Predation Medium	Fallout	<i>None</i>	0.6%	0.7%	0.9%	1.0%
			<i>Low</i>	1.2%	1.4%	1.7%	1.7%
			<i>Medium</i>	1.4%	1.6%	1.7%	1.9%
			<i>High</i>	1.6%	1.7%	1.9%	2.0%
	Predation High	Fallout	<i>None</i>	0.9%	1.0%	1.2%	1.4%
			<i>Low</i>	1.5%	1.7%	1.9%	2.0%
			<i>Medium</i>	1.7%	1.9%	2.0%	2.2%
			<i>High</i>	1.9%	2.1%	2.2%	2.4%

3.3.3 Management actions as combined scenarios

Predator control at 90% in the Northwest Kauai scenario produced an increase in lambda from 0.913 to 0.985. The benefit increased for the first 25 years before declining (Figure 3-4). The contribution to the population from predator proof fencing (predator eradication) increased over time then stabilized (Figure 3-5). Chick translocation produced a delayed contribution that grew with the population (Figure 3-6).

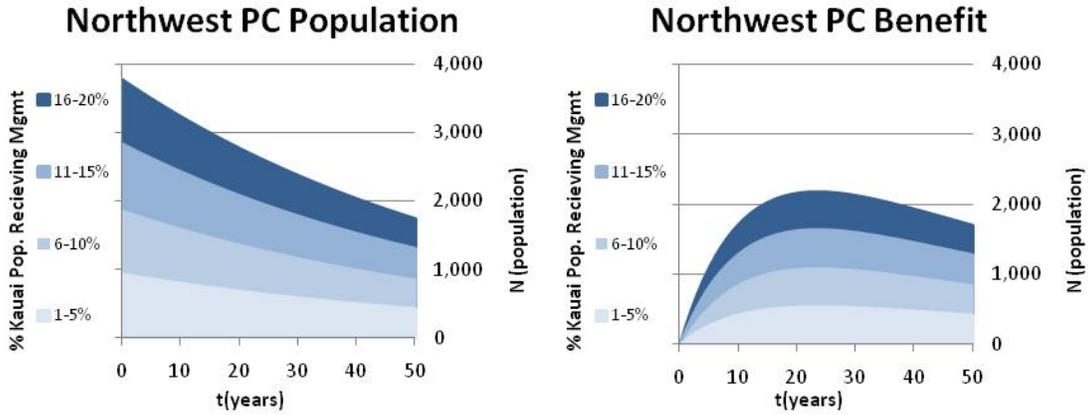


Figure 3-4 Predator control for 1-20% of Kauai population assuming low fallout with 0% SOS effectiveness and no powerline threats ($\lambda=0.985$, including a) growth rate of population receiving management, and b) benefit expressed as the number of birds added to the total population compared to no management ($\lambda=0.913$).

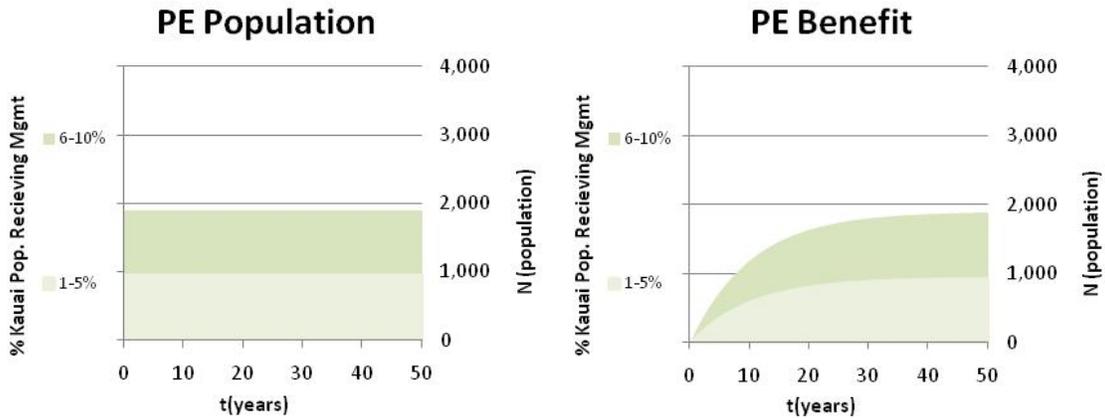


Figure 3-5 Predator eradication for 1-10% of Kauai population assuming no fallout and no powerline threats ($\lambda=1.000$), including a) growth rate of population receiving management, and b) benefit expressed as the number of birds added to the total population compared to no management ($\lambda=0.906$).

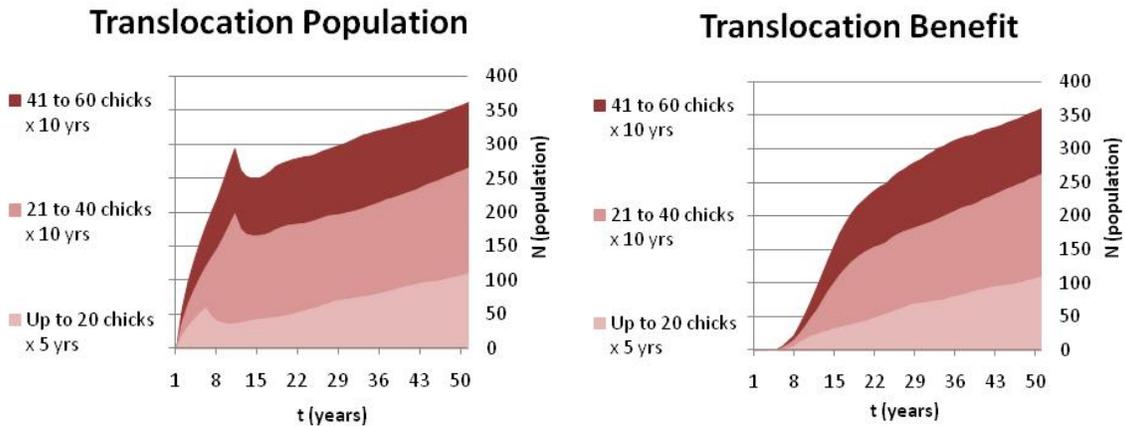


Figure 3-6 Newell's shearwaters in a translocated population of up to 20 chicks per year for five years, 21 to 40 chicks per year for 10 years, and 41 to 60 chicks per year for 10 years, including a) projected growth rate ($\lambda=1.012$), and b) translocation benefit expressed as the numbers of birds added to the total population compared to no management ($\lambda=0.906$).

3.3.3.1 Kauai population with management actions

Increasing the proportion of the population being actively managed decreased the rate of decline, and increased the remaining population in any given year (Figure 3-7). The initial trend of the total population resulting from these management combinations was ongoing decline because the overall population growth rate was a combination of a proportion of the population being managed and the remainder not being managed. However, the longer term trend for the Kauai population was toward stable because of the growth in predator proof fenced areas and realistic growth of translocated chicks.

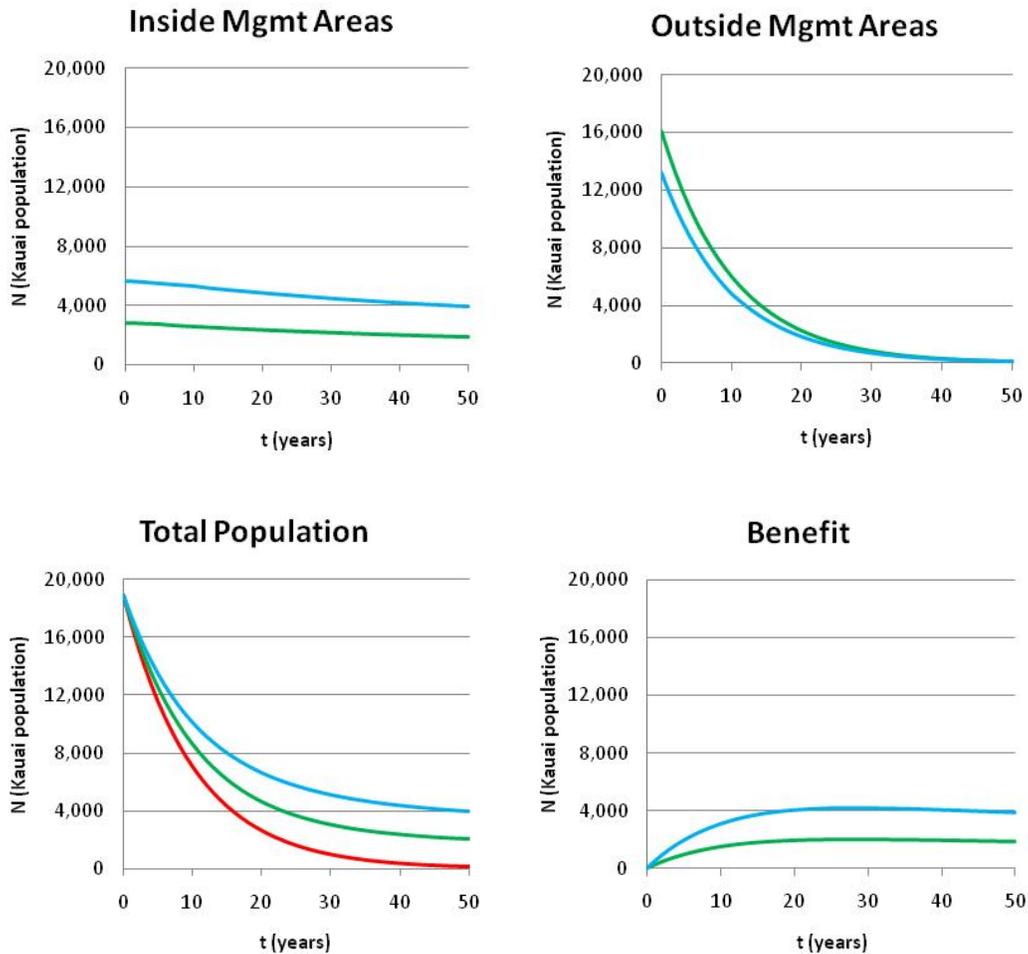


Figure 3-7

Total Kauai population with combined management actions, including a) growth rate inside management areas, b) growth rate outside management areas, c) growth rate for total (inside plus outside), and d) benefit expressed as the additional numbers of birds added to the total population. Red indicates baseline growth with no management actions. Green indicates management combination 1 with 10% population subjected to predator control, 5% to predator eradication, and 100 chicks translocated over five years. Blue indicates management combination 2 with 20% population subjected to predator control, 10% to predator eradication, and 400 chicks translocated over 10 years.

4 Discussion

Model outputs are a reflection of the quality of model inputs, and thus our results need to be interpreted accordingly. The absence of direct measures of management effectiveness for the actions we modeled are arguably the largest limitation for Newell's shearwater recovery and HCP planning. While positive effects in chick production and increasing the size of the adult population are expected outcomes from these actions, the absolute numbers presented here are artifacts of the modeling inputs and should not be considered absolute predictors of the future. They are best interpreted as population scale targets for mitigation and recovery planning.

4.1 Stable, Realistic and Optimal Growth

We provided a comprehensive assessment of vital rates for Procellariidae, and expect these to be appropriate for Newell's shearwaters under stable, growth and optimal conditions. Like other long-lived and low-fecundity K-selected species, Newell's shearwaters have limited capacity for population growth, making them vulnerable to even small reductions in their fecundity or survivorship (Warham 1990).

The strongest growth curve we could find in burrowing petrels was that for the Cahow or Bermuda petrel (*Pterodroma cahow*). The Cahow population size is estimated at 140 pairs and breeding is restricted to two islands, one of which has been used as a translocation site with the first recorded natural chick fledging there in 2009. Terrestrial threats are intensively controlled and management actions since the 1950's have included social attraction, chick translocation, artificial burrow creation, habitat restoration, removal of nest site competitors (i.e., *Phaethon lepturus*), and rat eradication. This population has no immigration or emigration. It appears free from terrestrial density dependent factors (i.e., nest sites) but does face some threats from light pollution, and possibly contaminants and hurricanes which damage breeding habitat and reduce breeding attempts (Birdlife 2010). Under these conditions the Cahow population grew from ~19 to ~80 breeding pairs in 45 years; a 3.2% mean annual growth rate (Figure 4-1) (Brooke et al. 2010). This growth rate is a function of the intense management directed at this species and likely lack of density-dependent factors bearing on use of foraging and breeding habitat that would trend a larger population towards stable (e.g. Short-tailed shearwaters in the Bass Strait, Tasmania). We do not consider this growth rate currently achievable for the Newell's shearwater population because, unlike management actions being implemented for the Cahow, management of the entire Newell's shearwater population is unlikely.

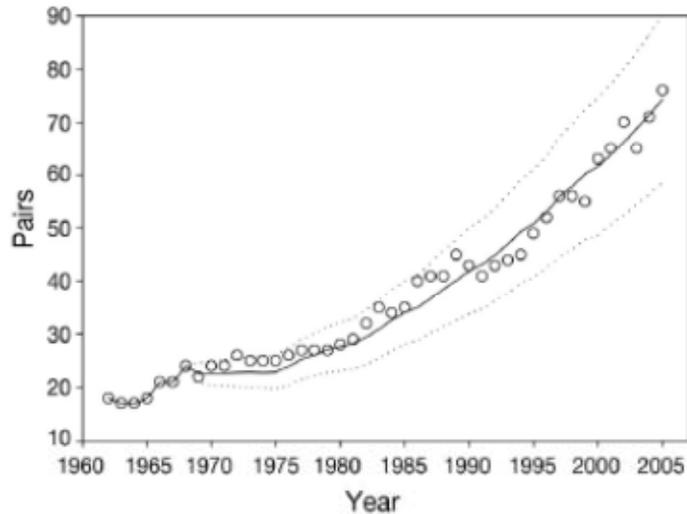


Fig. 4. *Pterodroma cahow*. Observed population trend (number of breeding pairs) for cahow between 1962 and 2005 (○), the predicted trend from the best-fit model (solid line) and the standard errors of the predictions (dotted lines). Note that the first 7 points in the series are used to seed the subsequent model predictions and so observed and predicted values are identical

Figure 4-1 *Pterodroma cahow* population trends from Brooke et al. (2010)

4.2 Threats to Newell's Shearwaters

4.2.1 Comparison of projected declines with Ornithological Radar and SOS trends

Ornithological Radar and Save Our Shearwaters data provide approximate measures of population trends for us to compare to the model results. These population indices are subject to caveats, particularly spatial limitations. If we assume that Radar and SOS trend data represent true population trends over time, these modeled data produce declines of 5.2 - 10.1% per year. Only when medium or high predation levels are included does lambda match the scale of decline suggested by Radar and SOS trends. Notably, the radar model using only data from the peak Newell's shearwater traffic period from 1993-2010 and the SOS model using data from 1988-2009 (Table 3-2, Appendix 3) indicated declines equal to or worse (0.899, 0.905) than the lowest lambda produced in our threat models (0.906). This worst case scenario was the best fit to these data trends.

4.2.2 Comparison to previous modeling efforts

Previous modeling by Ainley et al. (2001) may have underestimated the decline of the Newell's shearwater. These authors predicted annual declines of 3.2 to 6.1 % (λ of 0.968 to 0.939) based on their worst case scenario (high powerline mortality, predation and no SOS program). Our scenario that most closely matched the radar and SOS trends (high predation, high fallout and high powerline scenarios) indicates that a more severe decline of ~9 to 10% per year (0.906) was occurring during the last two decades.

The primary difference between our growth rates and those in Ainley et al. (2001) is the baseline from which adjustments for each threat were made. We chose a baseline with the assumption of a stable population without threats. Comparatively, the baseline in Ainley et al. (2001) indicates that if all identified threats were removed, the population would still be in decline, (i.e., not all threats were identified, or the severity of the threats was underestimated). We used higher mortality rates than did Ainley et al. (2001) based on the effect of predation on reproductive success and fallout. Powerline mortality estimates were the same. Both models are limited by the approach used, whereby growth rates are a function of adjusting a baseline population, as opposed to comparing a population with no threats to a population with threats (See Appendix 2).

4.2.3 What vital rate should management efforts target to maximize recovery?

Initial sensitivity statistics were consistent with other modeling studies of petrels (Martinez-Gomez and Jacobsen 2004, Oro et al. 2004) and k-selected vertebrates, i.e. management activities that reduce adult mortality will have the greatest impact on growth rate.

However, for Newell's shearwaters, the rate of fallout mortality (with 0% SOS effectiveness) produced a greater effect on overall growth rates than that of sub-adult and adult mortality from powerline strikes (declines of 0.6, 0.8 and 1% per year versus 0.1, 0.3, 0.4% per year, respectively). Thus, while adult mortality has a greater effect than reproductive success on growth rate when compared equally in a sensitivity analysis (Section 3.1.1), in this instance the *magnitude* of fledgling mortality from fallout outweighs the effect of sub-adult and adult deaths from powerline strike.

Consequently, targeting fallout minimization (and thus increasing reproductive success as a targeted vital rate) is clearly a worthwhile management strategy to increase growth rates of Newell's shearwaters, and from a practical standpoint, it is simple and inexpensive to achieve (i.e. turn out lights). We also highlight that fallout minimization must be undertaken in combination with powerline minimization (increasing adult survival as a vital rate), as either threat has the capacity to reduce growth rates below zero.

4.3 Management Scenarios

The management scenarios presented here have relevant application towards HCP and recovery goals. Recovery requires cessation of the decline (i.e. stable or growing population) and HCPs are required to offset the impact from proposed activities and concurrently not place the population in jeopardy (i.e. decline). While HCPs are not responsible for achieving recovery, management actions undertaken in a HCP should clearly contribute to recovery.

Fallout and powerline minimization will be critical to achieving recovery, thus to allow the population to have the best chance to stabilize in managed areas. Further, this would delay extirpation in unmanaged areas. At managed colonies, reduced predation mortality will increase the number of birds at risk to fallout and powerline strike without minimization efforts. Fallout minimization has increased on Kauai in the last five years (Erichsen and Griesemer 2009) and considerable minimization efforts are planned

under the KSHCP (KSHCP 2010). These planned KSHCP activities have considerable potential to aid in reducing the fallout and powerline mortality. Monitoring will prove critical to quantify effects.

Powerline minimization efforts are also planned (KIUC STHCP 2010). These plans are based upon recommendations by Ainley et al. (1995), who identified line characteristics presenting the highest threat; including lines >15.2 m in height, wide vertical arrays and lines located in valleys. Minimization efforts are currently planned with heavy emphasis on the east and south shores, where historically high numbers of birds have been recorded from easily accessible roadside locations. We note however that surveys have not been completed under lines that also meet these high risk criteria (height, array depth and location) but are not readily accessible, including inland transmission lines extending from Kahili to Hanalei River, and West to the Wainiha transmission station. Complete surveys (or other methods of assessing mortality) of all high risk line segments must be achieved to ensure that minimization is efficiently applied across all lines on the island, and not just those easily accessible.

For our combined management models, we assumed SOS provided no positive effect for the population as there is little evidence to the contrary. However, our investigation of the effect on growth rate if the SOS program has 100% effectiveness demonstrated great potential value, increasing growth rate up to 0.8% under high predation threats. The potential positive effect increased with higher levels of predation because the number of birds recovered by SOS represented a larger proportion of the depredated fledgling population. Theoretically, the program could significantly help slow decline in colonies outside management areas and compensate for additional required growth inside management areas. Therefore, studies should be conducted to evaluate the effectiveness of SOS (Ainley et al. 2001), given recent program improvement. Future investment should be made to bolster SOS rehabilitation methods and facilities (Linda Elliot, Hawaii Wildlife Center, pers. comm.) and release efforts.

Predator control activities are currently the most achievable colony management actions for larger proportions of the Newell's shearwater population, particularly in concert with watershed management projects (www.hawp.org). Colonies in Northwest Kauai present the greatest opportunity for predator control to achieve HCP and recovery goals because birds using these areas are likely subject to lower risk of powerline and fallout mortality, and thus allow for greater adult survivorship and chick production levels in the short-term. In our Northwest Kauai scenario, undertaking predator control had a significant effect of reducing the decline projected there, but a stable population was not reached because this population would still be experiencing 10% predation mortality and low fallout. We expect that 90% effectiveness from predator control is a realistic maximum given the difficulty of such actions in Kauai's mountainous terrain. An unknown component to predator control effectiveness will be rat control, and more information to determine the role of rat predation on this species is needed. If appropriate rat control cannot be provided, then there will need to be greater emphasis on eradication and translocation projects. The resulting benefit from predator control, or difference in the population over time compared to no management action, increased rapidly for the first 25 years but then steadily declined. Consequently, predator control as a sole management response represents a higher risk for recovery and HCPs when viewed in the long-term. This activity should be viewed in combination with other actions, and will play a crucial role in allowing a greater proportion of the population to persist for chick translocation and predator eradication projects.

A noteworthy example of the success of predator control and habitat management has been that of Hawaiian petrels at Haleakala National Park. In the early 80's, ongoing predation was identified as a key limiting threat to Hawaiian petrels, with annual adult survival as low as 0.80 in some years (Simons 1984, 1985). Ongoing predator control began in the park in the mid-1980s, focusing on removing cats, rats, and mongoose from the western crater rim, and fencing was completed in the late-1980s to stop burrow trampling by ungulates (Simons and Hodges 1998). Increased breeding success was subsequently reported (Hodges and Nagata 2001) and anecdotal observations suggest that the extent of used breeding habitat has expanded (Holmes et al. 2010). While Newell's shearwaters breed in markedly different terrain than Hawaiian petrels at Haleakala, this predator control example provides considerable optimism for similar projects with Newell's shearwaters, and suggests that population growth may be achievable under this management action alone.

Predator proof fencing and pest eradication represent an important opportunity to encourage Newell's shearwater recovery. Our predator eradication model had both zero fallout and powerline mortality, and allowed a stable population growth rate to be reached. Predator eradication for 10% of the population (with no fallout and no powerline mortality) provided approximately the same benefit as 20% of the population receiving predator control in our models (with low fallout and 10 % mortality) at 25 years. While no site currently exists to provide zero fallout risk to a predator eradication project, turning off sufficient artificial lighting to achieve this is certainly plausible, and fallout minimization will be critical to achieving required growth rates for these projects. Predator eradication projects on Kauai are also currently limited by toxin application guidelines to achieve rodent eradication, and the feasibility of implementing predator proof fences over Kauai's steep and rugged terrain. Overcoming these limitations should be considered a high priority for management, particularly given the broad ecosystem benefit these projects can have, and finding solutions to implement these projects should be included in HCP planning.

The translocation model provided a delayed benefit in comparison to predator control or eradication, with additional birds added to the population occurring when translocated chicks return as adults to breed for the first time (age six). These projected results are comparable to other translocation programs for Procellariidae. In 1999 and 2000, 100 Gould's petrel (*Pterodroma leucoptera leucoptera*) chicks per year were translocated from Cabbage Tree Island to Boondalbah Island in NSW Australia (Priddel and Carlile 2006). In 2009, more than 70 breeding pairs were estimated to be using the new breeding location (including additional natural recruitment), with more than 60 fledglings produced over the 8-9 years since chicks were first translocated (Priddel and Carlile 2009). As for the critically endangered Bermuda petrel, 79 translocated chicks fledged from Nonsuch Island during 2004-2008 (in addition to management actions of social attraction and more) with the first natural adult raised chick fledgling in 2009 (Birdlife International 2008, 2010b).

For our largest translocation scenario we deliberately chose low numbers of chicks moved, primarily because of the logistics necessary to translocate 600 chicks over 10 years. Subsequently, this scenario produced an additional 260 birds over 25 years, compared to protecting 5% of the population (with 90% predator control effectiveness, and no powerline, low fallout and predation mortality) from predators, which produced an additional 550 birds. However, in chick translocation models we projected a positive

growth rate because the population would be free of terrestrial threats. Further, the logistics to manage an appropriate threat-free environment (such as Kaena Point) would likely be markedly simpler (and possibly cheaper in the long-term) than ongoing predator control on a large scale on Kauai. For these reasons, chick translocation action should be considered fundamental to Newell's shearwater recovery and targeted by HCP planners.

Management actions need to operate in concert to achieve maximum potential benefit for areas most likely to grow. Minimization efforts should be prioritized to overlap in areas where colony-based management actions are planned, thus to maximize potential colony growth rates there. Notably, minimization plans for powerlines and fallout include considerable effort on the south and east shores (KIUC STHCP 2010). However, these actions may not prove to be the most effective because at least three colonies appear inactive there (Holmes et al. 2009), and fewer birds would benefit from these actions. Fallout and powerline minimization will aid in slowing the decline on the south and east shores, however, strategically, greater potential benefit would be achieved by first prioritizing minimization on the North Shore. The latter is where human densities are lowest, the most habitat appears to be available, and the greatest percentage of the population appears to exist.

Data used in this modeling effort was appropriate for an island-wide scale investigation; however regional differences in threat levels that we could not account for will play an important role in management planning. For example, high calling rates strongly indicate that cliff environments in the Na Pali region are also breeding habitat (KESRP, unpublished data) and they may hold significant numbers of Newell's shearwaters. The location poses little to no threat from lights and collision, and the topography may greatly reduce predation (either access for cats, or water for rats). These colonies represent potential refuges if colonies in other habitats continue to decline, and efforts should be made to determine the role and value of these habitats.

The management actions described here have clear potential to provide offset for incidental take of Newell's shearwaters. The two combined management scenarios modeled would provide a benefit of 2,000 and 4,000 birds over 25 years. However incidental take can only be approved should the population not be placed in jeopardy. Colonies with no management will likely still be in decline, but populations could be stable or growing in management areas. Sufficient allocation of resources to achieve stable or positive growth rates in managed areas will be critical to ensure that benefits can be sustainable in the long-term thus contributing to recovery.

Ultimately, a "recovered" (i.e. stable, self-sustaining) Newell's shearwater population in 25 years will likely be much reduced in size from what exists today. The size of that population will be a function of the amount of management provided, the speed at which it can be implemented, and the response of the colonies to that management. However, it is certainly realistic that subsets of the population can achieve positive growth rates when predation, fallout and powerline mortality are removed. Combined management actions likely to provide the greatest potential benefit should be targeted to prevent this species from becoming extinct, including predator control of Northwest colonies, predator eradication projects in combination with aggressive powerline and fallout minimization in the same region, and chick translocation to threat free environments.

5 Acknowledgements

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Appendix 1: Breeding success of Procellariiformes

Scientific Name	Colony Location	Vegetation	Non-native predators not recorded				Non-native predators present					Non-native predators eradicated or controlled					Reference	
			Min	Max	x	n	Predators	Min	Max	x	n	Mgmt action	Min	Max	x	n		
<i>Pachyptila belcheri</i>	Mayes I., Kerguelen I.				0.52	14												Nevoux and Barbraud 2006
<i>Pagodroma nivea</i>	Adelie, Antarctica	arctic	0.21	0.80	0.51	27												Chastel et al. 1993
<i>Puffinus griseus</i>	Tuhawaiki I., NZ				0.63	3												Cuthbert & Davis 2002 (Ibis)
<i>P. griseus</i>	Snares I., NZ	dense shrub (fern)	0.20	0.52	0.35	3												Newman et al. 2009
<i>P. griseus</i>	Whenua Hou I., NZ	dense shrub (fern)	0.70	0.82	0.76	6												Newman et al. 2009
<i>P.mauretanicus</i>	Sa Cella I., Med. Sea	coastal cliffs	0.59	0.62	0.61	2												Oro et al. 2004
<i>P.puffinus</i>	Rhum I., Scotland	ridges	0.44	0.64	0.52	2												Brooke 1990
<i>P. puffinus</i>	Rhum I., Scotland	ridges			0.58	14												Thompson and Furness 1991
<i>P. puffinus</i>	Skokholm I., Wales	grasses, open canopy	0.62	0.75	0.70	4												Brooke 1990
<i>P. puffinus</i>	Canna I., Scotland				0.62	13												Thompson and Furness 1991
<i>P.tenuirostris</i>	Fisher I., AU		0.25	0.83	0.61	≥40												Wooller et al. 1989
<i>P.carneipes</i>	Woody I., AU						rats, ravens	0.15	0.66	0.42	2							Powell et al. 2007
<i>P.carneipes</i>	Lord Howe I., AU						rats			0.51	1							Priddel et al. 2006
<i>P.iherminieri</i>	Santa Cruz I., Galapagos						rats			0.22	3							Harris 1969 in Finkelstein 2010
<i>Pachyptila belcheri</i>	New I., Falkland Is.	lowland					rats, mice, cats	0.39	0.83	0.63	7							Catry et al. 2007
<i>P.belcheri</i>	New I., Falkland Is.	tussock					mice and cats (rats CO)	0.09	0.58	0.37	3	CO			0.80	1		Catry et al. 2007
<i>Procellaria aequinoctialis</i>	Crozet I., Possession I., S. Indian Ocean						rats					CO			0.51	9		Jouventin et al. 2003 in ACAP2009
<i>P.cinerea</i>	Macquarie I., Australia						rats only (cats ER)				0.46	4						ACAP 2009b
<i>P.cinerea</i>	Marion I., Prince Edward Is., S. Africa						cats	0.00	0.50	0.13	3							Newton and Fugler 1989 in ACAP 2009b

<i>P. parkinsoni</i>	Little Barrier Island, NZ		rats, cats	0.14	0.83	0.45	8							Imber 1987
<i>P.parkinsoni</i>	Great Barrier Island, NZ		cats, rats, pigs					CO	0.69	0.84	0.76	9		Bell et al. 2007
<i>P.westlandica</i>	Punakaiki, NZ		cats, stoats, rats, dogs					CO			0.62	7 (95-02)		Waugh et al. 2006 in ACAP 2009c
<i>Calonectris diomedea</i>	Congreso I., Spain	veg.slopes, rocky cliffs	rats	0.27	0.44	0.36	2	CO	0.51	0.71	0.67	5		Igual et al. 2006
<i>Pterodroma cahow</i>	Bermuda		rats	0.28	0.66		NA	CO	0.50	0.60		NA		Wingate 1978 in Carlile et al. 2003
<i>P. leucoptera</i>	Cabbage Tree I., New Zealand	Dense forest	avian and weeds (CO)	0.23	0.26	0.25	3	CO			0.31	1		Priddel et al. 1995
<i>P. phaeopygia</i>	Floreana I., Galapagos Is., Ecuador	dormant volcanic	cats, rats, pigs, goats, donkeys		0.31	NA	CO	0.23		0.72	0.53	4		Cruz and Cruz 1996
<i>P.sandwichensis</i>	Maui, Hawaii, USA	dense shrub and semi-open canopy	cats, rats			0.39	1	CO	0.61	0.71	0.66	2		Simons 1985
<i>P.sandwichensis</i>	Maui, Hawaii, USA	dense shrub and semi-open canopy	cats, rats	0.00	0.44	0.23	8	CO	0.17	0.50	0.38	8		Natividad Hodges and Nagata 2001
<i>Puffinus huttoni</i>	South I., NZ	alpine, shrub, open canopy	stoats					CO	0.27	0.66	0.47	9		Cuthbert & Davis 2002 (Ibis)
<i>P.newelli</i>	Kauai, Hawaii, USA	dense shrub, semi-open canopy	cats, rats, barn owls					CO	0.39	0.75	0.59	5		Telfer 1986
<i>P. yelkouan</i>	Port-Cros I., Med.Sea	coastal cliffs and boulders	cats					CO			0.73	3		Bonnaud et al. 2009
<i>P.assimilis</i>	Chickens I., NZ	coastal cliffs	rats			0.20	2	ER			0.75	2		Parrish 2005
<i>P.assimilis</i>	Lady Alice I., NZ		rats	0.38	0.50	0.43	2	ER	0.52	0.86	0.68	7		Pierce 2002
<i>P.assimilis</i>	Coppermine I., NZ		rats	0.05	0.06	0.05	5	ER	0.38	0.79	0.56	3		Pierce 2002
<i>P. mauretanicus</i>	Conills I., Med. Sea	coastal cliffs	rats					ER	0.48	0.75	0.60	2		Oro et al. 2004
<i>P.opisthomelas</i>	Natividad I., Mexico		cats			0.36	2	ER						Keitt 2002
<i>Pterodroma cookii</i>	Codfish I., New Zealand (high study area)	dense, forested ridges	rats	0.00	0.69	0.32	16	ER	0.54	0.88	0.77	3		Imber et al 2003
<i>P. cookii</i>	Codfish I., New Zealand (low study area)		rats	0.50	0.85	0.66	16	ER	0.57	0.78	0.72	3		Imber et al 2003
<i>P. cookii</i>	Little Barrier Island, NZ		cats, rats			0.32	6	ER			0.59	3		Rayner et al. 2007

<i>P. cookii</i>	Little Barrier Island, NZ		rats only (cats ER)	0.09	6	ER					Rayner et al. 2007		
<i>P. macroptera gouldi</i>	Whale I., New Zealand		rats, rabbits	0.00	0.07	6	ER	0.25	0.52	9	Imber 1976		
<i>P. macroptera macroptera</i>	Marion I. (sub-Antarctic island)	grasses, mosses	cats	0.00	0.21	0.16	5	ER		0.64	1	Cooper et al. 1994	
<i>P. pycrofti</i>	Chickens Is., NZ		rats			0.20	2	ER		0.75	2	Parrish 2005	
<i>P. pycrofti</i>	Lady Alice I., NZ		rats	0.19	0.51	0.35	2	ER	0.52	0.68	0.60	7	Pierce 2002
<i>P. pycrofti</i>	Coppermine I., NZ		rats	0.22	0.37	0.31	5	ER	0.48	0.71	0.59	3	Pierce 2002
<i>Procellaria aequinoctialis</i>	Marion I. (sub-Antarctic island)		cats			0.36	1	ER		0.22	1	Cooper et al. 1994	
<i>Halobaena caerulea</i>	Marion I. (sub-Antarctic island)		cats			0.24	1	ER		0.64	1	Cooper et al. 1994	
<i>Calonectris diomedea</i>	Selvagem Grande I., Portugal (Area ABCD)		rabbits, mice			0.37	13	ER		0.54	5	Zino et al. 2008	
<i>C. diomedea</i>	Selvagem Grande I., Portugal (Area EFGH)		rabbits, mice			0.45	13	ER		0.56	5	Zino et al. 2008	

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Appendix 2: Ainley’s Threat Models

Ainley et al. (2001) measures of predation, fallout, and powerline threats recalculated in a Lefkovitch model are presented in Table 1 and Figure 1.

We noted differences in our results in comparison to age-based (Leslie) matrices (and age-based population modeling software such as Vortex) using the same inputs but with an maximum age of 36 years (as used in Ainley et al. (2001)). In general, Lefkovitch models produced increasingly higher growth rates with decreasing λ values when compared to Leslie models and using life history values reported in Ainley et al. (2001), likely because Lefkovitch models allow some individuals to live beyond 36 years.

Table 1 Growth rates for Newell’s shearwater under low medium and high projected levels of fallout, powerline and predation threats, using values from Ainley et al..

				Powerline			
				<i>None</i>	<i>Low</i>	<i>Medium</i>	<i>High</i>
Predation <i>None</i>	Fallout	<i>None</i>		0.975	0.974	0.972	0.971
		<i>Low</i>		0.967	0.965	0.964	0.962
		<i>Medium</i>		0.965	0.963	0.962	0.960
		<i>High</i>		0.962	0.961	0.960	0.958
Predation <i>Present</i>	Fallout	<i>None</i>		0.952	0.948	0.949	0.948
		<i>Low</i>		0.945	0.944	0.942	0.941
		<i>Medium</i>		0.943	0.942	0.940	0.939
		<i>High</i>		0.941	0.940	0.939	0.937

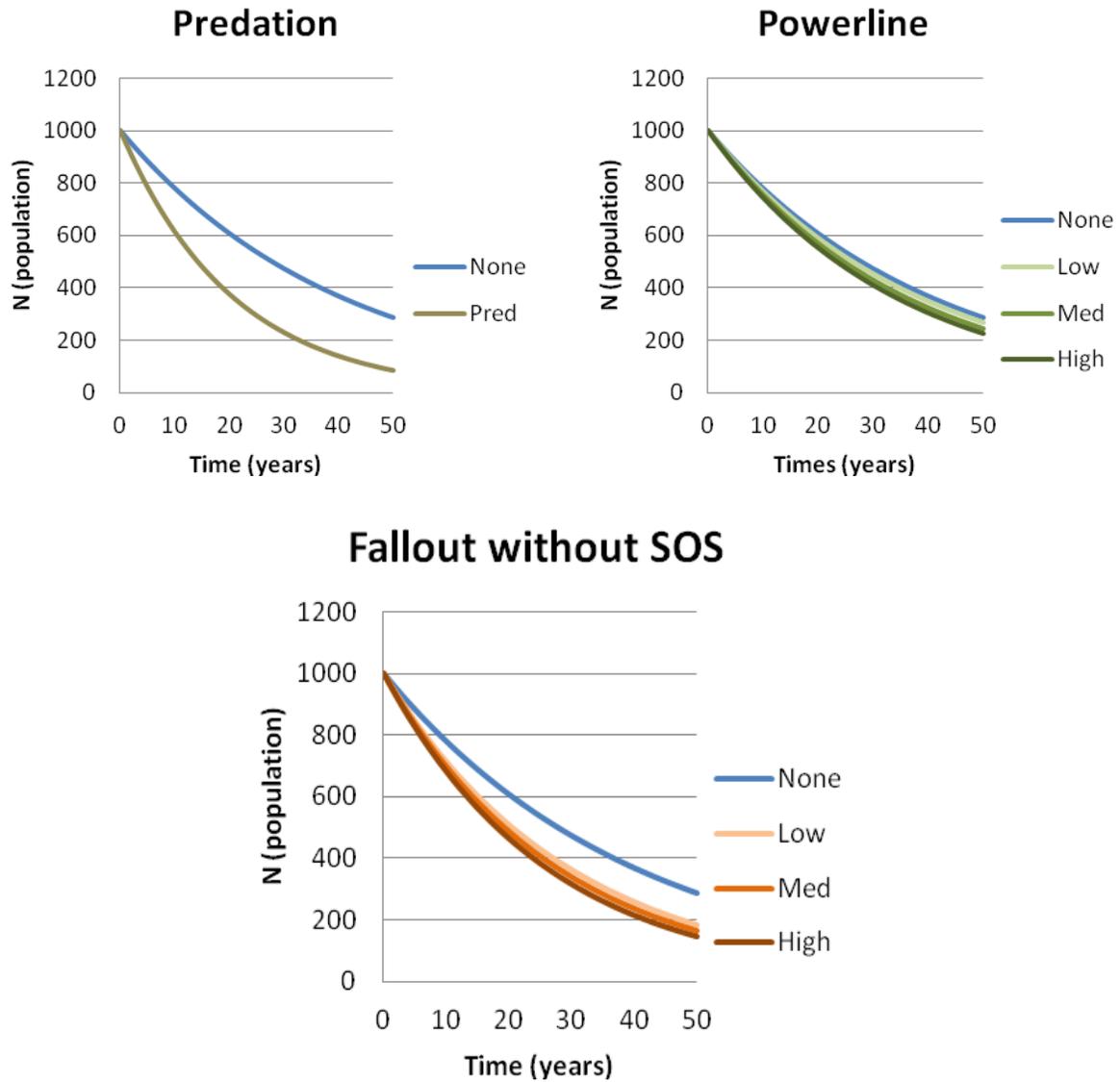


Figure 1 Effects of predation, powerline and fallout threats (assuming 0% SOS effectiveness or Ainley’s “without SOS”) on growth rate, using Ainley’s baseline vital rates and threat adjustments.

Appendix 2 References

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Appendix 3: Save Our Shearwater and Ornithological Radar Trends on Kauai, Hawaii, 1993-2010

Ornithological Radar

Ornithological radar is a monitoring tool used to determine an annual population index of endangered seabirds on Kauai, and other cryptic seabirds such as the marbled murrelet *Brachyramphus marmoratus* (Raphael et al. 2002). This tool offers observation independent of light conditions (allowing observations at night) and over a large spatial scale (3 km).

Ornithological radar data has been collected on Kauai in 1993 and 1999-2001 by Day et al. (2003), and between 2004-2009 by the State of Hawaii. The radar study initiated in 1993 was designed to document the spatial variation in passage rates of Hawaiian petrels (*Pterodroma sandwichensis*) and Newell's shearwaters (*Puffinus newelli*) transiting from the ocean to breeding colonies to identify regions of highest traffic. Since 1999, the following survey design has been implemented as a proxy monitoring tool. Thirteen sites are surveyed during June, a period during which breeders are incubating and non-breeders are present in the colony. Each site was surveyed for a single night for 2-3 hours in June, beginning at 7pm (approximately 18-26 min prior to sunset), during a high traffic period when birds returned to colonies after spending the day foraging at sea (Cooper and Day 1998).

Radar Data

Complete description of field methods can be found in Cooper and Day (1998), Day et al. (2003) and Holmes (2006). From 2004-2010 a Furuno 1510 MARK-3 12 kW X-band marine surveillance radar was used. The 2 meter (m) array was mounted ~2 m above ground level and tilted ~15° to reduce ground clutter. Radar was set to a 1.5 km radius and 0.07 μ/second pulse length. Data were expressed as a number of targets per hour.

Following Day et al. (2003) radar data was filtered to remove targets flying less than 30 mph. We included targets flying 25 mph for the site of Hanalei because Hawaiian petrels and Newell's shearwaters were visibly seen gaining elevation, and considered flying slower at this site. Data were filtered to include birds only flying inland based on direction (i.e. birds flying outbound, or only over the ocean, were not included). We removed targets classified as very small by operators (likely bug traffic, particularly on evenings with wind >10 mph).

Because we only had access to summary data for 1993 and 1999-2001, we filtered and organized data from 2004 onwards to match these earlier data conditions. This included 4 x 30 min sessions beginning at 7pm. A disadvantage of this approach is that species arrival (an important consideration in separating species) is dependent on sunset time, which in turn changes by 8 min over the course of June. A more appropriate data organization would be to sort data into sessions beginning at sunset, which we recommend for future analyses.

From 2006 onwards timing of radar data collection was reduced from 3 hours after sunset to two, because a smaller percentage of data was being collected in this final hour and made little difference to the overall value for that evening. Data from 2005 and earlier were similarly filtered.

Cooper and Day (1998) identified that species-specific differences in the arrival patterns of Hawaiian petrels and Newell's shearwaters and used this to identify species specific trends. Hawaiian petrels were identified as flying inland from 0-60 min after sunset and as Newell's shearwaters from 30 min after sunset onwards. Day et al. (2003) subsequently considered their session 1 (18-26 min before sunset – 12-4 min after sunset) likely to include primarily Hawaiian petrels, and their session 3-onwards (34-42 min after sunset onwards) to be primarily Newell's shearwaters only. A disadvantage of this approach is that it requires disregarding the data from the highest traffic period (session 2) when both species were recorded.

A potential source of error in Kauai radar data include changes in the spatial (countable) area of the radar screen, either from survey sites being moved to new locations (e.g. new housing developments forcing survey site to be moved) or changes in ground clutter evident on the screen (e.g. growth in trees since Hurricane Iniki blocking new areas of the screen). We consider these to be minor and would not significantly change broad trends evident in the dataset.

We ran four models using radar data from 1993-2010. The first two including all radar data and data from 8pm onwards [(i.e. suggested to be primarily Newell's shearwaters only following Day et al. (2003)]. Because the 1993 value was such an outlier (twice as high as that collected in 1999-2001), we also ran these two models again without 1993 values.

Save Our Shearwater population trends

Attraction or confusion of nocturnal Procellariiformes to artificial lights is a well-documented phenomenon around the world and often results in birds colliding with structures or exhausting themselves, (Hailman 1979, Le Corre et al. 2002, Telfer et al. 1987). "Fallout" is the general term for the grounding of seabirds due to artificial night light (Reed et al. 1986, Telfer et al. 1987). Each year on Kauai, 100's of Newell's shearwater fledglings experience disorientation due to artificial lights during their maiden flight to sea, and are killed in collisions with artificial structures, or by dogs, cars, and cats when they land on the ground. Between 1979-2009 more than 30,000 grounded Newell's shearwaters fledglings have been collected as part of the Save Our Shearwater (SOS) program (State of Hawaii unpublished data), a conservation program that encourages the public to rescue fallout birds with the goal of minimizing mortality of grounded birds. The public is encouraged via television, radio, and print media to rescue any downed shearwaters and deliver them to aid stations, where they are picked up by SOS staff and given a second chance to fledge to the ocean.

The Save Our Shearwater dataset also has become a proxy monitoring tool but there are several significant practical and statistical limitations when the data are used to this end. These include, but are not limited to problems associated with a lack of standardized search effort by the general public, a bias towards collecting live birds, predation of live grounded birds by cats dogs (and not being identified by the public), being run over by cars (and not identifiable by the public), and decomposition and

scavenging of dead birds (Ainley et al. 2001). These considerations mean the number of Newell's shearwaters recovered by SOS is a minimum number, and has been estimated up to only represent half of true fallout (Ainley et al. 2001).

SOS Data

Data were filtered to only include fledgling Newell's shearwaters collected under the SOS program each year. It is likely some adults were misidentified as fledglings during the SOS season, although we considered this to represent few birds and unlikely to have significant impact on the modeling outcomes.

Two major trends are evident in the number of Newell's shearwaters recovered by SOS. First, a period of no trend from 1979-1991, albeit with high year-to-year variation, and second, a period of decline from 1987/1988-2009. We fit two models to SOS data from 1979-2009 and 1988-2009 to cover these two periods.

Data analyses

We investigated the Ornithological Radar trends over time by fitting a linear mixed model (REML) of Radar targets / per hour by Year. Radar site was classified site as a random variable (245 data points) and year as a repeated term. We investigated SOS data over time by fitting a simple linear model to the number of fallout fledgling Newell's shearwaters over time (Year). Fallout and Radar data were log transformed prior to analyses.

For all models, model diagnostics were checked using standard graphical methods (Quinn and Keogh 2002). We present the parameter estimate and 95% Confidence Intervals for Year provided a measure population change each year in these data, which we used to compare to growth rates (λ) in subsequent population modeling. Adjusted R^2 is presented to provide an indication of how much variation is explained by the parameters used in each model. Data analyses undertaken using SAS JMP.

Results and Discussion.

All SOS and radar models showed a steep decline with a mean annual decline from to -0.052 ($\lambda=0.948$) to -0.107 ($\lambda=0.899$). Radar data including 1993 values showed steeper declines (-0.085, -0.095) than . when the 1993 was excluded (-0.052, -0.77). Data from the period suggested to be primarily Newell's shearwaters (8-9pm) showed steeper declines (-0.095 from 1993-2010, -0.077 from 1999-2010) than all radar data combined.

Fallout data from 1988-2009 showed a decline of 0.101 per year. When fallout data from 1979-1991 (a period with no trend evident in the raw data) was included in the SOS model a strong decline is still evident (-0.070 per year). .

Adjusted R^2 values suggested that 62-91% of the variation in the models were explained by year (sos) or year and site (radar) alone. These outcomes offer a subjective comparison to population modeling results in this effort, and we offer the same caveats in interpretation provided in the methods.

Table 1 Model terms and outcomes for fitting models of SOS and Ornithological Radar date by year.

Model	Years	Adjusted R ²	1 + Parameter estimated for the term year (λ)	Lower 95% CI	Upper 95% CI
Radar13.ALL. 93-10	1993-2010	.77	0.915	0.901	0.928
Radar13.3&4.93-10	1993-2010	.63	0.905	0.888	0.922
Radar13.ALL. 99-10	1999-2010	.75	0.948	0.927	0.968
Radar13.3&4.99-10	1999-2010	.62	0.923	0.898	0.947
SOS.79-09	1979-2009	.81	0.930	0.917	0.942
SOS.88-09	1988-2009	.91	0.899	0.885	0.914

Appendix 3 References

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Appendix 4: Kalaheo and KPNWR Newell's Shearwater Data

Table 1 Status and fate of Newell's shearwaters at Kalaheo Kauai, 1981-1985 (Telfer 1986)

	1981	1982	1983	1984	1985	mean
Number of previously active burrows checked¹	36	39	47	44	38	40.8
Number of active burrows (breeding)	22	24	14	17	16	18.6
<i>Percent burrows previously active with eggs laid</i>	61.1%	61.5%	29.8%	38.6%	42.1%	46.6%
Nests known to have failed	9	17	4	7	4	8.2
<i>Percent nests failed</i>	40.9%	70.8%	28.6%	41.2%	25.0%	41.3%
Nests known to have produced fledglings	13	7	10	10	12	10.4
<i>Reproductive success</i>	59.1%	29.2%	71.4%	58.8%	75.0%	59%²
Causes identified for Nest failures	Owls? Rats	Feral cats	Hurricane Iwa	Rats	Rats and Cats	

1. These burrows were previously active during 1979-1980 .
2. Ainley et al. (1995, 2001) report Telfer's (1986) breeding success mean at 66%

Table 0 Status and fate of Newell's shearwaters at Kalaheo Kauai, 1993-1994 (Ainley et al. 1995)

	1993	1994	mean
Number of burrows checked	58 ¹	58	
Number of burrows with activity (breeding and non-breeding)	42	33	
Number of burrows with breeders	15 ²	16	
<i>Percent burrows w/activity with eggs laid</i>	35.7%	48.5%	42.1%
Nests known to have failed	11	3	
<i>Percent nests failed</i>	73.3%	18.8%	46.0%
Nests known to have produced fledglings	4	13	
<i>Reproductive success</i>	26.7% ³	81.3%	54.0%

1. An additional 7 burrow like structures were investigated but not included in analyses
2. 16 burrows with activity were reported in 1993 in Table 2, and 15 reported twice in the text (p18, 19). We use 15 in our analyses because it most closely matched Ainley et al.'s (1995) reported reproductive success measures.
3. Only 7% breeding success is reported for 1993 (p.18) but we consider this a typographic error.

Table 3 Status and fate of 3 Newell’s shearwater pairs from Kilauea Point National Wildlife Refuge, since year of discovery (Zaun 2007).

Year	B2	B6	B3	Combined
1997	fledged chick			
1998	failed ¹			
1999	fledged chick			
2000	fledged chick			
2001	fledged chick	fledged chick		
2002	fledged chick	fledged chick		
2003	fledged chick	fledged chick		
2004	fledged chick	fledged chick		
2005	fledged chick	fledged chick		
2006	fledged chick	fledged chick		
2007	fledged chick	fledged chick		
2008	fledged chick	fledged chick	fledged chick	
2009	fledged chick	fledged chick	fledged chick	
Number of breeding attempts	12	9	2	23
Breeding probability	100%	100%	100%	100%
Breeding success	100%	100%	100%	100%

1. In 1998, the adults and egg from this nest were moved to an artificial next box, and the breeding attempt was abandoned. We did not include this in our measure of reproductive success.

Appendix 4 References

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Appendix 5: Adult survival and mass in Procellariiformes

Species	Sx	Sx Reference ¹	Mass (g) Dunning (2008) ²
<i>Hydrobates pelagicus</i>	0.880		25
<i>Oceanites oceanicus</i>	0.910		32
<i>Oceanodroma leucorhoa</i>	0.860		37
<i>Bulweria bulweria</i>	0.950		99
<i>Pachyptila turtur</i>	0.840		137
<i>Pelecanoides urinatrix</i>	0.667		141
<i>Pagodroma nivea</i>	0.934	Chastel et al. 1993	268
<i>Puffinus huttoni</i>	0.931	Cuthbert and Davis 2002	364
<i>Daption capense</i>	0.930		425
<i>Daption capense</i>	0.950		425
<i>Pterodroma sandiwichensis</i>	0.930		434
<i>Puffinus puffinus</i>	0.905	Brooke 1990	454
<i>Calonectris diomedea</i>	0.950		535
<i>Calonectris diomedea</i>	0.927	Mougin et al 2000	535
<i>Puffinus tenuirostris</i>	0.920	Skira 1991	559
<i>Puffinus tenuirostris</i>	0.912	Hunter et al. 2000	559
<i>Fulmarus glacialis</i>	0.970		613
<i>Puffinus griseus</i>	0.930		787
<i>Puffinus griseus</i>	0.917	Clucas et al. 2008	787
<i>Puffinus griseus</i>	0.952	Clucas et al. 2008	787
<i>Puffinus griseus</i>	0.966	Clucas et al. 2008	787
<i>Puffinus griseus</i>	0.909	Richdale in Scofield et al. 2001	787
<i>Fulmarus glaciodes</i>	0.900		795
<i>Fulmarus glaciodes</i>	0.950		795
<i>Procellaria westlandica</i>	0.965	Waugh et al 2006	1199
<i>Procellaria aequinoctialis</i>	0.895	Barbraud et al. 2008	1213
<i>Thalassarche chlororhynchos</i>	0.910		2200
<i>Phoebastria fusca</i>	0.950		2585
<i>Thalassarche bulleri</i>	0.890		2950
<i>Thalassarche bulleri</i>	0.950	Sagar et al. 2000	2950
<i>Phoebastria immutabilis</i>	0.950		3150
<i>Phoebastria palpebrata</i>	0.970		3150
<i>Phoebastria irrorata</i>	0.960		3395
<i>Thalassarche chrysotoma</i>	0.950		3508
<i>Thalassarche melanophrys</i>	0.880		3564
<i>Thalassarche melanophrys</i>	0.930		3564
<i>Macronectes halli</i>	0.900		4185
<i>Macronectes giganteus</i>	0.900		4395
<i>Diomedea exulans</i>	0.970		8190
<i>Diomedea epomorpha</i>	0.940		9000
<i>Diomedea epomorpha</i>	0.970		9000

1. All references from Warham 1996 unless stated otherwise
2. Where mass was presented for both male and female we use the calculated mean

Appendix 5 References

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Appendix 6: Fallout Mortality Rates with 100% Save Our Shearwaters Program Effectiveness

To investigate the *theoretical* range of the effect of the Save Our Shearwaters (SOS) program on population growth rates, we modeled that fledglings recovered and released through SOS experience equivalent survival to fledglings not experiencing fallout (or 100% SOS effectiveness).

We estimated the mortality of the 280 to 560 downed fledglings by combining the discovery levels (Section 2.4.3.2) with morbidity values of 7.7%, 15%, 25%, and 43% based on Ainley et al. (2001). Fallout fledgling mortality ranged between 22 and 241 dead fledglings (Table 1), and produces 64 mortality rates for each level of predation (Table 2).

To model fallout mortality given the large number of possible mortality rates, we selected the 17%, 50%, 83% percentile of the range of values at each predation level to represent low, medium, and high fallout assuming 100% SOS effectiveness (Table 3). For consistency in modeling threats as rate adjustments, we present the *decrease in reproductive success rates* at each level (Table 3) of fledgling population size rather than the *proportional decreases in reproductive success* (Table 2).

Table 1 Newell’s shearwater fledgling fallout mortality (100% SOS effectiveness), based on four levels of discovery and morbidity and a population decline of 75% from Spear et al. (1995).

Morbidity	Discovery Rate			
	1.00	0.80	0.67	0.50
0.077	22	27	32	43
0.150	42	53	63	84
0.250	70	88	104	140
0.430	120	151	180	241

Table 2

Newell's shearwater fledgling fallout mortality rates (in the presence of SOS), based on four levels of predation, as a proportion of total annual chick production and decline of 75% from Spear et al. (1995).

Predation	Morbidity	Discovery Rate			
		1	0.8	0.67	0.5
none	0.077	0.007	0.009	0.011	0.015
	0.150	0.015	0.018	0.022	0.029
	0.250	0.024	0.030	0.036	0.048
	0.430	0.042	0.052	0.062	0.083
low	0.077	0.010	0.013	0.015	0.020
	0.150	0.020	0.025	0.030	0.040
	0.250	0.033	0.042	0.050	0.066
	0.430	0.057	0.071	0.085	0.114
medium	0.077	0.017	0.021	0.025	0.034
	0.150	0.033	0.042	0.050	0.066
	0.250	0.055	0.069	0.083	0.111
	0.430	0.095	0.119	0.142	0.190
high	0.077	0.036	0.045	0.053	0.072
	0.150	0.070	0.087	0.104	0.140
	0.250	0.116	0.145	0.174	0.233
	0.430	0.200	0.250	0.299	0.400

Table 3

Newell's shearwater reproductive success adjustments for fledgling fallout mortality used to investigate effect of SOS management action

	Fallout Threat level	No. of dead fledglings	Percentile of value range	Predation			
				none	Low	medium	High
Fallout with SOS program (100% effectiveness)	Low	58	0.17	-0.012	-0.014	-0.016	-0.019
	Medium	131	0.50	-0.027	-0.031	-0.036	-0.044
	High	204	0.83	-0.042	-0.048	-0.057	-0.068

Appendix 6 References

Ainley DG, Podolsky R, Deforest L, Spencer G, Nur N. 2001. The status and population trends of the Newell's Shearwater on Kaua'i: Insights from modeling. *Studies in Avian Biology No 22 Evolution, Ecology, Conservation, and Management of Hawaiian Birds: A Vanishing Avifauna*:108-123.