

Effectiveness and cost-effectiveness of yellow-eyed penguin restoration treatments

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Abstract

*Although an estimated US\$6 billion is invested annually in our planet's biological diversity, little research has been conducted on which conservation treatments work best or provide best value for money. Where controlled experiments are not possible, econometric techniques can be used to determine the effectiveness of conservation treatments. We use a long-running yellow-eyed penguin (*Megadyptes antipodes*) nest count in New Zealand to compare the effectiveness and cost effectiveness of three commonly used endangered species recovery treatments—trapping of introduced predators, revegetation, and intensive management. Following ecological theory, we specify a density-dependent population growth rate. We control for year effects and site characteristics such as land cover, slope, and elevation. The possibility of selection bias in treatment is confronted with site fixed effects and with an instrumental variable based on site accessibility. Of the three treatments that we analyze, only intensive management is significantly correlated with increases in annual site-level yellow-eyed penguin population growth rate. We estimate that intensive management increased the yellow-eyed penguin population by 9% above the counterfactual, and that the average cost of producing an additional yellow-eyed penguin nest through intensive management is NZ\$68,600.*

Keywords: Yellow-eyed penguin, conservation effectiveness, conservation cost-effectiveness, predator control, revegetation, intensive management

Introduction

Conserving our planet's biological diversity is a grand investment. Recent estimates of annual conservation expenditures include US\$6 billion on nature reserves worldwide (James *et al*, 1999), US\$1.5 billion by international conservation organizations (Halpern *et al*, 2006), and NZ\$106.5 million on management of natural heritage in New Zealand (DOC, 2004). Despite the magnitude of this financial outlay, little research has been conducted on which conservation investments are most successful, or provide the best value for money. The Millennium Ecosystem Assessment laments that "few well designed empirical analyses assess even the most common biodiversity conservation measures" (MEA, 2005, p.122). Ferraro and Pattanayak (2006, p.482) suggest that "if any progress is to be made in stemming the global decline of biodiversity, the field of conservation policy must adopt state-of-the-art program evaluation methods to determine what works and when." Knowing the rate of return on different conservation treatments would enable conservationists to direct scarce resources to the most effective treatments (Wilson *et al*, 2007).

A small but growing body of economic literature has evaluated the effectiveness of conservation programs. Conservation program effectiveness is the improvement in biological outcome attributable to the program, per amount of treatment applied. Ferraro *et al* (2007) found that species listed under the U.S. Endangered Species Act showed recovery relative to comparable non-listed species only if their listing was accompanied by funding. Bruner *et al* (2001) found that land within tropical protected areas lost less forest cover than adjacent land outside protected areas. Sanchez-Azofeifa (2007) and

Sills *et al* (forthcoming) examine the effectiveness of Costa Rica's Payments for Environmental Services program in deterring deforestation. Other efforts to determine effectiveness of conservation treatments have been hindered when objective metrics of conservation effort or biological outcome do not exist (Abbitt and Scott, 2001), or when these data have not been collected (Kiesecker *et al*, 2007).

Some research has gone a step further, calculating the cost effectiveness of conservation programs. Conservation program cost effectiveness is the improvement in biological outcome attributable to the program, per dollar spent. Shwiff *et al* (2005) compare the cost effectiveness of predator removal and monitoring for the endangered least tern at Camp Pendleton. Engeman *et al* (2002) compare the cost effectiveness of four predator control methods for protecting endangered sea turtles. Cullen *et al* (2001; 2005) study the cost effectiveness of single species and multiple species conservation programs in New Zealand.

Ideally, the effectiveness of a conservation treatment can be tested through a controlled ecological experiment such as a before-after, control-impact, paired (BACIP) study (Stewart-Oaten *et al*, 1986). However, a BACIP study must be carefully planned in advance of providing conservation treatment; such a study can't be performed after a treatment regime is already in place. Next best is a study of treatments that have been randomly assigned across sites. But conservation treatments are often applied in a deliberately non-random fashion. Treatments may be applied to sites where they are expected to be most successful, or to sites which are most easily accessible.

When a BACIP study is infeasible, and treatments have not been randomly assigned, panel econometric techniques can be used to determine the effectiveness of

conservation treatments. In this paper we apply these techniques to a long-running recovery program for the endangered yellow-eyed penguin (*Megadyptes antipodes*) to evaluate the effectiveness of three common endangered species recovery treatments—trapping of introduced predators, revegetation, and intensive management. We use penguin nest counts that span 15 years and all 48 South Island nesting sites, and exploit cross-sectional and longitudinal variation in the application of the three treatments, to determine the effectiveness of each treatment. We combine effectiveness results with indicative data on treatment costs to estimate the cost-effectiveness of each treatment as well.

The yellow-eyed penguin

The yellow-eyed penguin, or hoiho, is the third largest penguin. It stands up to 65-70 cm and weighs up to 6 kg. It is recognizable by a distinctive yellow eye band and pupil. The yellow-eyed penguin is endemic to New Zealand where its range is restricted to Stewart Island, Campbell Island, Auckland Island, and the southeast coast of the South Island. The yellow-eyed penguin feeds in the ocean and nests on coastal land. In prehuman times the yellow-eyed penguin nested in coastal forest and shrub margins (Marchant and Higgins, 1990), though since the arrival of humans most of this native forest has been replaced by stocked pasture.

The yellow-eyed penguin is a long-lived species, attaining 20 years or more (Richdale, 1957). Females begin breeding at 2-3 years, while males begin breeding at 2-5 years (Marchant and Higgins, 1990). Yellow-eyed penguins are philopatric, meaning

that they generally return to the nest area of their birth to breed. Richdale (1957) found that 81% of birds returned to their nest area or to a nearby area to breed. Darby (1996) found that close to 90% of birds bred within 500 meters of their nest area. Once they have chosen a nest site, adult yellow-eyed penguins have extremely high nest site fidelity (McKinlay, 2001). A review of yellow-eyed penguin movements over 15 years showed that only 14 of 2999 adult birds moved from one established breeding area to another (Darby, 1996). Acceptable nests must have a protected back and must be visually isolated from other nesting pairs (Seddon and Davis, 1989). Yellow-eyed penguins typically occupy nest sites in July, begin breeding in late August and early September, and lay eggs in September. Up to two chicks from each nest hatch in early November and fledge in early February (Darby and Seddon 1990). After fledging, chicks become juveniles, who head out to sea with no further parental supervision (Seddon, 1990). Mean juvenile mortality has been estimated at 52% (Richdale, 1957), but has been recorded as high as 88% at one site (Darby and Seddon, 1990).

A substantial nature tourism industry has developed around viewing the yellow-eyed penguin. An estimated 126,000 tourists, or 5.7% of New Zealand's 2.2 million international visitors in 2006-2007, viewed penguins while in New Zealand, though this figure also includes tourists who viewed blue penguins and Fiordland crested penguins (Ministry of Tourism, 2007 a). With the average holiday visitor spending NZ\$3115 in New Zealand (Ministry of Tourism, 2007b), penguins attract millions of dollars to the New Zealand economy. On a local level, Tisdell (2007) estimates that wildlife viewing on the Otago Peninsula, where the yellow-eyed penguin and royal albatross are flagship species, generates NZ\$6.5 million in direct revenue and NZ\$100 million in flow-on

expenditure annually. Visitors to yellow-eyed penguin viewing sites report feelings of wonder, improved mood, and increased environmental awareness (Schanzel and McIntosh, 2000).

The yellow-eyed penguin faces a variety of threats. On land, chicks face predation from mustelids (ferrets and stoats), cats, and dogs, while juveniles and adults face predation only from dogs. At sea, juvenile and adult yellow-eyed penguins are vulnerable to sea lions, sharks, and gill nets. Starvation, trauma, and disease also contribute to penguin mortality (Hocken, 2005). Toxic algal blooms have been responsible for penguin mortality (Shumway *et al*, 2003). A scrub fire killed over 60 adult yellow-eyed penguins at Te Rere Reserve in February 1995 (Taylor, 2000, p.22). Unmanaged tourism can negatively impact yellow-eyed penguin reproductive success and juvenile survival (McClung *et al*, 2003; Ellenberg *et al*, 2007). Nesting sites are always vulnerable to further habitat loss (McKinlay, 2001).

The IUCN Red List classifies the yellow-eyed penguin as 'endangered' due to its small breeding range, declining habitat quality, and extreme fluctuations in its population (Birdlife International, 2007a). The yellow-eyed penguin is one of the three most endangered penguin species, along with the Galapagos Penguin (Birdlife International 2007b) and the Erect-Crested Penguin (Birdlife International, 2007c). It is one of seventy critically endangered, endangered, or vulnerable bird species in New Zealand (IUCN, 2007). The New Zealand Department of Conservation (DOC) classifies the yellow-eyed penguin as 'nationally vulnerable' (Hitchmough *et al*, 2005) due to its restricted range and steep declines over portions of this range in the recent past (McKinlay, 2001). Yellow-eyed penguins numbered an estimated 5930 -6970 birds in 1997 (McKinlay,

2001). However, yellow-eyed penguin population may never have been high (Moore 2001), and Ratz (1997) has shown that during the period 1959-94 no overall decline occurred in yellow-eyed penguin numbers on the Otago Peninsula. The 2007 IUCN Red List assessment suggests that the yellow-eyed penguin might be downlisted in the future as a result of ongoing conservation measures (Birdlife International, 2007a). It is the goal of DOC's Hoiho Recovery Plan to increase South Island yellow-eyed penguin nests from 458 in 2000 to 1000 by 2025 (McKinlay 2001). By the 2006 breeding season there were 464 yellow-eyed penguin nests on the South Island (DOC unpublished).

The yellow-eyed penguin recovery effort has attracted a mosaic of conservationists. The New Zealand Department of Conservation, the private Yellow Eyed Penguin Trust, and individual landowners and conservationists are all contributing effort towards the recovery of the species. These actors have implemented a diverse range of yellow-eyed penguin recovery treatments across sites and years. This paper calculates the effects of three recovery treatments—trapping of introduced predators, revegetation, and intensive management—on annual site-level yellow-eyed penguin population growth rate

Trapping

Yellow-eyed penguins, like other New Zealand birds, evolved in the absence of terrestrial mammalian predators. When humans brought terrestrial mammals to New Zealand, bird populations of many species were decimated or driven to extinction (O'Donnell, 1996). Today, non-native ferrets, stoats, cats, and dogs all contribute to the

terrestrial mortality of yellow-eyed penguins (Hocken, 2005). To reduce terrestrial mortality of chicks, a common yellow-eyed penguin recovery treatment is trapping of mustelids and feral cats. Diverse trapping methods have been used across years and sites. At some sites poison-baited traps are set in lines across a property; at other sites traps are placed near known penguin nests. At some sites traps are placed and checked at intervals throughout the year; at other sites traps are placed and maintained only during the nesting season (MacFarlane, personal communication, 2007). This analysis does not distinguish among these styles of trapping predators.

Revegetation

The native forest that once formed the terrestrial habitat for yellow-eyed penguins has been greatly reduced since human settlement. Today large areas of the southeast coast of the South Island are in pasture, with small patches of remnant scrub and forest cover. Pasture is considered a less hospitable nesting environment for yellow-eyed penguins than taller vegetation because direct sunlight may result in hyperthermia. Penguins require cool, shaded conditions, enclosed nests, and possibly visual isolation to breed successfully (Seddon and Davis, 1990). To increase breeding success, another common yellow-eyed penguin recovery treatment is revegetation. Diverse revegetation methods have been applied. At some sites, all grassland is seeded with native trees and shrubs, and weeded several times in the following years until native bush can take hold. At other sites, constructed nest boxes are placed in the pastoral landscape and surrounded

by stands of flax (*Phormium tenax*) (MacFarlane, personal communication, 2007). This analysis does not distinguish between these two styles of revegetation.

Intensive management

Yellow-eyed penguins of all age classes are impacted by disease, starvation, and trauma. To reduce these impacts, full time managers provide intensive management at some sites. These managers regularly check the status of individual penguins. Managers provide sick penguins with antibiotics, injured penguins with medical care, and underweight penguins with food supplements. This treatment is provided to penguins collected on-site and to penguins brought in from elsewhere. In addition to providing treatment for individual birds, managers can enhance trapping by placing traps near nests and monitoring these traps more frequently than would otherwise be possible. Managers can also maintain nest boxes. Furthermore, managers can enhance revegetation efforts by ensuring that trees are cared for once planted (Ratz, personal communication, 2007).

Data

Our dependent variable, annual site-level population growth rate, is constructed from the New Zealand Department of Conservation's panel data set of yellow-eyed penguin nest counts. These counts span all 48 nest sites along a 300 km stretch of the southeast coast of New Zealand's South Island from 1992-1993 to 2006-2007 (DOC unpublished data). This is a continuation of work pioneered by John Darby and others

(Seddon *et al*, 1989). Each site was visited by volunteers a minimum of three times during the breeding season, with the number of occupied nests estimated after the three visits (McKinlay, personal communication, 2007). Occupied nest boxes are considered equivalent to occupied nests. A survey of yellow-eyed penguin population data from Banks Peninsula across six consecutive breeding seasons and six sites was not included in our analysis because this survey counted eggs, adults, and chicks rather than nests.

Each occupied nest represents one breeding pair of adult yellow-eyed penguins. Nest counts are a reasonable proxy for total adult population size; in most years 60-80% of adult yellow-eyed penguins breed, though the percentage is lower during exceptionally bad years (Efford *et al*, 1994). The total yellow-eyed penguin population at each site is not used as a metric because juvenile mortality is high and variable. The total adult population was not used as a metric because this is more difficult to survey than nests.

A panel data set of the three yellow-eyed penguin recovery treatments was compiled across sites and years based on conversations with practitioners in the field (R. Goldsworthy, D. MacFarlane, B. McKinlay, C. Lalas, H. Ratz, and F. Sutherland, pers.comm). Analysis was limited to terrestrial management actions because these actions are easily observable and site-specific. The treatment variable in a site-year was coded as one if that treatment was applied during that site-year, and was coded as zero otherwise. Some sites never received any treatment; other sites received a particular treatment in all years. The treatment variable switched from zero to one at many sites, but never switched from one to zero; see Figure 1. A Venn diagram of site-years receiving each combination of treatments is shown in Figure 2.

Ecological theory predicts that site-level population growth rate should be dependent on population density. Data on site area (McKinlay 1997) were used to calculate nest density. Available site characteristics were compiled for use as control and instrumental variables. Data on penguin site locations from MapToaster TopoNZ 2007 were used to calculate distance from Dunedin, the largest city in the region, using Google Maps. Data on site land cover, slope, aspect, and distance from road are taken from the New Zealand Land Cover Database, which has a minimum mapping unit of one hectare (Terralink, 2007). The percent of land cover in each vegetation type was determined for a 250m radius about the center of each site using GIS. Summary statistics are shown in Table 1.

The cost per hectare of implementing each treatment was estimated following discussions with practitioners (B. McKinlay, A. Spencer, R. Goldsworthy personal communication, 2007); see Table 2. These figures represent crude estimates of what it would have cost DOC to apply a treatment at a site in a given year. Disaggregated expenditure records were not available. Actual expenditure by private groups on revegetation and intensive management was likely lower than the cost that would have been incurred by DOC, as these groups were able to use volunteer rather than paid labor. DOC replacement costs are used in this analysis to be conservative and consistent across treatments. We recognize that volunteers can in some instances provide labor inputs more cheaply while receiving welfare-improving psychic benefits. Overhead costs, for instance office costs and vehicle costs, were not included in the cost per hectare, as it is assumed that these costs would have been borne whether or not a particular yellow-eyed penguin recovery treatment was applied to a particular site.

Method

In the simplest model, a population's stochastic, logarithmic population growth rate is dependent upon treatments, density, and year effects:

$$\ln l_{it} = b_0 + X_{it}'b_1 + b_2 \ln d_{it-1} + b_3 y_t + e_{it} \quad (1)$$

Here the observation $l_{it} = n_t/n_{t-1}$ represents the annual population growth rate in nests, n , at site i between year $t-1$ and t . X_{it} is a matrix of dichotomous variables representing whether or not each treatment was employed at site i in year t . d_{t-1} represents nest density (nests per hectare) at site i in year $t-1$. y_t is a year dummy. A timeline of treatments and nest counts is shown in Figure 3. All regressions were performed using Stata, and are OLS unless otherwise noted.

Population growth rate, rather than nest numbers or absolute change in nest numbers, is the proper dependent variable for several reasons. First, population growth rate l_t is multiplicative rather than additive. l_t is expected to be insensitive to number of nests in the previous period, n_{t-1} , while absolute change in nest numbers, $n_t - n_{t-1}$, is not. Stewart-Oaten *et al* (1986) confirm that the correct parameter of interest is the mean of the underlying probabilistic process that produces abundance, rather than abundance itself. Second, nest numbers are susceptible to undercounting. It is likely that undercounting is more severe at sites which are more remote, more vegetated, steeper, or otherwise more difficult to survey (McKinlay, personal communication, 2007). By

assuming that undercounting of nests occurs proportionally in both n_t and n_{t-1} , the dependent variable population growth rate neutralizes heterogeneity in measurement error across sites. If nest counts at a site are conducted more thoroughly after a treatment is put in place than before, this would introduce a small amount of bias by making treatments appear more favourable during the first year of treatment only. Nest counting occurred with greater frequency in treated site-years (263/272; 96.7%) than in untreated site-years (383/448; 85.5%). Positive nest numbers were recorded with greater frequency in treated site-years (239/263; 90.9%) than in untreated site-years (301/383; 78.6%; See Table 1). The natural log of l_t is used as the dependent variable rather than l_t because the mean of $\ln l_t$ is symmetric around zero for periods of time in which growth rate is zero. Observations were included only when both n_t and n_{t-1} were counted and positive, leaving 519 usable observations. Nests dropped to zero and did not return to positive the following year at only three sites. Of these sites, two were untreated and one was treated with trapping and revegetation. Thus removal of observations due to zero nests does not appear to be correlated with any particular treatment.

Ecological theory predicts that growth rate should diminish at a site as the population size approaches carrying capacity. To account for this effect, log of density was included as a regressor. This follows the ecological model of density-dependent population growth, $\frac{dN}{dt} = rN(1 - \frac{N}{K})$ (Lotka, 1925). Before accounting for density dependence, no treatment was significantly correlated with an increase in population growth rate; see results in Table 3(a). After accounting for density dependence, intensive management was significantly correlated with an increase in population growth rate; see Table 3(b). Density itself has a negative and highly significant effect on growth rate

throughout all regressions ; see Table 3(b-h). When year dummies were introduced to account for interannual fluctuations in penguin mortality and fecundity , explanatory power increased, and intensive management remained significant; see Table 3(c).

Temporal Effects

Potentially the treatment effect may not be visible immediately. The length of time a treatment has been in place could influence the treatment's effect on growth rate. For revegetation, higher nest densities have been recorded in dense, low cover of scrub mosaics and early stage regenerating forest than in relatively open understory of mature forest (Seddon, personal communication, 2007). So it could be the case that revegetation increases growth rate during initial or middle years, but decreases growth rate in later years. For trapping, it could be the case that trapping increases growth rate in early years as the threat from predators is reduced, but has no effect on growth rate once a new, predator-free, equilibrium is reached.

Delays of more than fifteen years in treatment effects can not be determined using our data. However, three alternative models are run to test for shorter-term time-variant treatment effects. First, the treatment regressors in the basic model are lagged by three years—the length of time for newborn chicks to enter the adult breeding population:

$$\ln l_{it} = b_0 + X_{it-3}'b_1 + b_2 \ln d_{it-1} + b_3 y_t + e_{it} \quad (2)$$

The significance of treatments is robust to a three year lag; see Table 3(d). Lags of other time lengths had the same result and are not presented. Second, the data was sub-sampled into progressively greater time intervals. The log of growth rate was regressed on explanatory and control variables, over k -year time intervals, where $k \geq 2$:

$$\ln l_{it} = b_0 + \sum_{j=1}^k X_{it-j} b_1 + b_2 d_{it-k} + b_3 y_t + e_{it} \quad (3)$$

Intensive management remained significant over every sub-sampled time interval but one, while trapping and revegetation were not significant in any time interval (see Table 4). Finally, treatments were interacted with the length of time they had been in place (1-5 years/6-10 years/11-15 years/16+ years). These interactions were not significant, and are not presented.

Omitted Variables

Another endangered species, the New Zealand (Hooker's) sea lion (*Phocarctos hookeri*), is only beginning to recolonize the South Island of New Zealand after being extirpated over a century ago. A single New Zealand sea lion is known to prey upon yellow-eyed penguins at two intensively managed sites (Lalas *et al*, 2007). The presence of the sea lion at this particular beach is considered exogenous to the choice of management. Since it is known that no other penguin beach was home to this sea lion, a regression is included that accounts for the presence of this sea lion. As expected, the presence of the sea lion is significantly correlated with a reduction in the population

growth rate. The magnitude of the intensive management coefficient increases after accounting for the sea lion; see Table 3(e).

Other omitted variables are likely to impact penguin population. Fire is known to have **or** negatively impacted penguin populations in one site-year (Taylor, 2000, p22), and is included as a control variable; see Table 3(f). Vandalism is also known to have negatively impacted penguin populations (McKinlay, personal communication, 2007), but because there is no complete record across sites and years of this activity, vandalism is not included as a control variable. If omitted variables are correlated with a particular treatment, then the effect of that treatment could be biased, though we know of no omitted variable for which this is the case. Trapping and revegetation are expected to be more robust to omitted variable bias than intensive management because these treatments occurred across more site-years.

Observable site characteristics were included as control variables. These included regional dummies (North Otago, Otago Peninsula, Catlins), percent of site in each land cover type (sand and gravel, grassland, scrub and shrubland, and forest), mean slope, mean elevation, distance from Dunedin, and distance from a road. There was no significant regional variation in the effect of treatments. No site attribute had a significant effect on growth rate, though this could be because the New Zealand Land Cover Database's minimum mapping unit of one hectare was potentially not fine enough to detect local landscape diversity. Both the sea lion and the fire had a negative and significant impact on growth rate. The significance of intensive management and insignificance of trapping and revegetation was robust to the inclusion of control variables; see Table 3(f). Interactions between treatments were included in an additional

regression and were not significant; see Table 3(g). Interactions between treatments and good or bad years were not significant and are not presented.

Spatial Effects

Population change at the site level is the most natural dependent variable because pre-breeding yellow-eyed penguins are largely philopatric and breeding yellow-eyed penguins exhibit extremely high nest site fidelity. However, some pre-breeding yellow-eyed penguins do move between sites, and the penguins' predators are mobile as well. Furthermore, intensive management could conceivably have a negative impact at nearby sites if injured birds are collected disproportionately from these sites, and these birds do not return after their rehabilitation. Sophisticated spatial autocorrelation techniques that could be used to test the impacts of treatments at nearby sites were precluded in this study by small sample size. A basic test was conducted for the possibility that the growth rate at one site could be influenced by treatments implemented at nearby sites:

$$\ln l_{it} = b_0 + X_{it}'b_1 + W_{it-1}'b_2 + b_3 \ln d_{it-1} + b_4 y_t + e_{it} \quad (4)$$

Here matrix W_{it} is made up three vectors, each representing the number of other sites in year t within 10 km of site i at which one of the three treatments was implemented. In this specification the effects of nearby treatments were not significant, and intensive management retained significance; see Table 3(h). When the presence of the sea lion was removed, the effect of intensive management on growth rate fell just below significance.

Confronting selection bias

Recovery treatments were not assigned randomly across site-years. It is conceivable that site characteristics could be correlated with both probability of treatment and penguin population growth rate or probability of success. If recovery treatments were more likely to be applied at sites where growth rate was most steeply decreasing, or where treatments would be least successful, then regression coefficients for the effect of treatments would be biased downward. If recovery treatments were more likely to be applied at sites where growth rate was most steeply increasing, or where treatments would be most successful, then regression coefficients for the effect of treatments would be biased upward.

There is no reason to think that treatments were preferentially applied to sites due to growing or declining populations. However, treatments might have been directed to sites where they were thought to have the highest probability of success, which is unobservable. Our discussions with practitioners indicate that locations for treatments were likely to be chosen based on where landowners were receptive to penguin management, or where land was up for sale at a particular time. These site characteristics were not observed. Still, some portion of treatment location appears to have been chosen based on observable, exogenous site characteristics. Treatments were disproportionately directed to sites nearer the population center of Dunedin, to sites where land cover was conducive to access and management, and to sites where penguins were already plentiful; see Table 1.

A two-stage least squares estimation was attempted to account for selection bias. Because there were three treatments, at least three instrumental variables were required. We considered variables that were correlated with the probability of treatment being applied at a site, but were plausibly uncorrelated with penguin growth rate and probability of treatment success. These variables included site distance from the major city of Dunedin, percent of site in grassland rather than shrubland or forest, mean slope and mean elevation at a site, and initial penguin population. Sites closer to Dunedin are more accessible for conservation, but are not expected to have higher or lower penguin growth rates or probability of success. Grassy sites are more conducive to management activities than sites covered in dense scrub or forest. Grassy sites might be expected to support a higher or lower nest density than scrub or forest sites, but would not be intrinsically expected to have higher or lower growth rates. However, grassy sites might be correlated with greater or lower probability of treatment success, so this was not included as an instrument. Flatter, lower sites may be more accessible than steeply sloping sites, but would not be expected to have higher or lower growth rates. It is plausible that the steepness of a site would not affect the success probability of treatments. Greater initial penguin population at a site might have made treatment more likely. While population size and treatment are endogenous, penguin population at a site during the earliest year of nest counts would not be affected by treatments at that site that occurred later.

The instrumental variables chosen were distance from Dunedin, mean slope, and initial population. These instruments are strong for trapping and intensive management ($f\text{-statistic} > 10$), but weak for revegetation ($f\text{-statistic} < 10$); see first stage results in Table

5. No treatment is significant in the two-stage least squares regression; see Table 6. A Hausman test shows that the f-statistic on the first stage residuals in the OLS regression is 1.26; see Table 6. Therefore we can not reject the null hypothesis that treatments are exogenous, and there is no reason to reject the OLS results in favour of the 2SLS results.

Next, a site fixed effects model was run. In the fixed effects model, the treatment effect was no longer identified from differences in growth rate across sites, but only from differences within sites across years. Effects of treatments that occurred prior to our data set were not captured. Figure 1 shows that there were eleven sites with intertemporal variation in trapping, and ten sites with intertemporal variation in revegetation, but only one site with intertemporal variation in intensive management. So, the fixed effects estimator is more reliable for trapping and revegetation than for intensive management. All fixed effects identification was based on the addition, rather than removal, of treatments. In the fixed effects regression, intensive management retained its magnitude, but was no longer significant. Trapping and revegetation remained insignificant; see Table 3(h).

Neither the two-stage least squares estimation nor the fixed effects model are completely reliable given the limitations of even this relatively large data set. The spectre of selection bias can not be eliminated entirely. This should be viewed as an argument for conservation managers to undertake BACIP studies, or randomize treatments across available sites, to allow an analysis free from potential selection bias.

Cost effectiveness

Ideally, cost-effectiveness could be compared across all three treatments, with the most cost-effective treatment being the preferred management strategy . But since intensive management was the only effective treatment, it was the only cost-effective treatment as well . However, computing the cost per additional nest obtained through intensive management is still useful to compare investments in this treatment with alternative uses of conservation resources.

In the second stage of analysis, the average cost-effectiveness of each recovery treatment was computed, using the following formula:

$$CE_x = \frac{N_{2006} - \hat{N}_{x2006}}{C_x} \quad (5)$$

Here CE_x is the average number of additional nests gained by 2006 per dollar spent on treatment x . N_{2006} is the actual number of nests across all sites in 2006. \hat{N}_{x2006} is the counterfactual number of nests in 2006, that is, the number of nests that would have been present in 2006 if treatment x had not been applied anywhere from 1992 -2006. Actual number of nests, N_{2006} , is equal to $\sum_i n_{i2006}$. Where n_{it} was not directly counted , its value was predicted using the inverse of the specification in (2):

$$n_{it} = n_{it-1} e^{\hat{b}_0 + X_{it}\hat{b}_1 + \hat{b}_2 \ln d_{it-1} + \hat{b}_3 y_t} \quad (6)$$

The result, $N_{2006} = 462.5$, is very close to DOC's estimate of $N_{2006} = 464.0$, which was obtained using the prediction model $n_{it} = 0.95n_{it-1}$ (McKinlay, personal communication, 2007.).

The counterfactual number of nests had no treatment been applied, \hat{N}_{x2006} , is equal to $\sum_i \hat{n}_{xi2006}$. When treatment x had not been applied at a site, counterfactual nests were equal to observed nests, $\hat{n}_{xit} = n_{it}$. When the treatment had been applied to a site, \hat{n}_{xit} was predicted by subtracting the influence of the treatment from the actual growth rate observed during a particular site-year, using the model:

$$\hat{n}_{xit} = n_{it-1} e^{\ln I_{it} - \hat{b}_x x_{it}} \quad (7)$$

Here, \hat{b}_x is the coefficient in the vector of coefficients \hat{b}_1 representing the effect of treatment x . Though density is a significant determinant of growth rate, it is not included in the model in (7) to prevent counterfactual population from being driven entirely by density. To create a confidence interval around \hat{N}_{x2006} at the 95% confidence level, the model in (7) was used, but $\hat{b}_{x.025}$ was substituted for \hat{b}_x to create an upper bound, and $\hat{b}_{x.975}$ was substituted for \hat{b}_x to create a lower bound.

C_x represents the total cost of treatment x from 1992-2006. As discussed in Data, C_x is the estimated cost had the treatment been supplied by DOC, rather than actual expenditure on the treatment over this time period. This analysis relied upon indicative estimates of costs rather than actual disaggregated expenditure records to determine

treatment cost effectiveness. Treatment costs were given in 2007 dollars and so were not discounted to present value. At the 95% confidence interval, neither trapping nor revegetation produced an additional yellow-eyed penguin nest at a finite cost. We find that the average cost of producing an additional yellow-eyed penguin nest through intensive management was NZ\$68,600; see Table 7.

Results

Of the three yellow-eyed penguin recovery treatments analyzed, only intensive management was significantly correlated with an increase in site-level yellow-eyed penguin population growth rate. This finding was robust to model specification, to inclusion of control variables, and to alternative tests for time-variant treatment effects over the 15-year study period. In a simple test for spatial spillover in treatments, the significance of intensive management was sensitive to the inclusion of a dummy to account for a sea lion. To confront potential selection bias, we attempted both fixed effects and two-stage least squares estimation. In the fixed effects estimation, intensive management retained its magnitude but not its significance, likely because intertemporal variation in intensive management occurred at only one site. In a two-stage least squares estimation using an instrument constructed from variables related to site accessibility and initial population, intensive management was not significant. However we could not reject the hypothesis that treatments were exogenous, so the two-stage least squares approach is not preferred to the ordinary least squares estimation. Trapping and revegetation were not significant in any model.

We find that intensive management was responsible for a 0.0523 average increase in log growth rate, or a 5.4% average increase in growth rate; see Table 3(c). We estimate that if no intensive management had been applied anywhere, there would have been 424 yellow-eyed penguin nests in 2006, rather than 462. Intensive management was responsible for 38 additional yellow-eyed penguin nests by 2006—a 9% increase from the counterfactual. This is equivalent to 0.67 average additional nests per site-year of intensive management, or NZ\$68,600 per additional nest; see Table 8(a).

At least two explanations could explain the greater population growth observed in intensively managed site-years. One explanation is that treating sick or injured adults directly decreases adult mortality. That decreased adult mortality would contribute to increased growth rates is consistent with two yellow-eyed penguin population viability analyses. McKinlay (1997) found that minimal improvements in the rate of adult mortality dramatically reduced the probability of extinction. Efford and Edge (1998) found that for penguins, like other long-lived seabirds, the population growth rate is particularly sensitive to changes in the adult survival rate. Treating sick, injured, or underweight chicks may indirectly decrease adult mortality as well, by allowing adult penguins to devote more food energy to themselves rather than to their chicks. This is especially true for the treatment of breeding females, and during low food years (Ratz, personal communication, 2007).

An alternative explanation for increased growth rate in intensively managed site-years is that penguins brought to intensively managed sites for hospitalization could be becoming habituated and choosing to remain at these new sites (Seddon, personal communication, 2007). As such, intensively managed sites may be acting as a partial

sink of penguins relocated from elsewhere rather than a source of new breeding adults. Further research into the movement patterns of hospitalized penguins could determine what aspect of intensive management is responsible for increasing nest numbers, and to what extent these nest numbers augment rather than replace nesting activity elsewhere.

Neither trapping nor revegetation was correlated with an increase in yellow-eyed penguin population growth rate. This may be because neither of these treatments directly decreases adult mortality, known to be important for yellow-eyed penguin population growth rates from the two PVAs mentioned above (McKinlay, 1997; Efford and Edge, 1998). Data on the particular style or the level of intensity with which treatments were implemented across sites was not collected. So, we can not rule out that one style or level of intensity of trapping or revegetation was correlated with increase in growth rate, but that this effect was diluted in the regression results by being pooled with less effective methods of trapping or revegetation.

That intensive management is correlated with increased population growth but trapping and revegetation are not is also consistent with an examination of 124 opportunistically collected yellow-eyed penguin specimens that found that penguin deaths caused by mustelids (5%) were exceeded by deaths caused by trauma (23%), natural causes including disease (16%), starvation (13%), marine predators (9%), dogs (8%), and drowning (7%) (Hocken, 2005). Intensive management has the potential to reduce mortality from trauma, disease, and starvation, which combined account for over half of the penguin deaths recorded in the Hocken study, while mustelid trapping has the potential to reduce a much smaller proportion of mortality.

Revegetation and trapping are not correlated with increased yellow-eyed penguin population growth rate, but these activities may provide ancillary benefits. Revegetation can bring the aesthetic and cultural benefits of restored native forest to a region where such forest is scarce. A WWF-New Zealand commissioned report (Buchan, 2007) explains that revegetation projects in New Zealand have benefits for participants outside of any biodiversity benefits, such as teaching nursery skills and building social capital. Trapping of introduced predators may benefit other bird species at sites where trapping occurs. Studies have found that predator control increased breeding success for the kaka (Moorhouse *et al*, 2003), the kokako (Innes *et al*, 1999), and the mohua (O'Donnell *et al*, 1996). Conservation programs are often multifaceted and are likely to be evaluated on more than just their success in producing penguins.

The presence of a single New Zealand sea lion had a large and significant negative impact on penguin growth rate at the two sites where it was present; see Table 3(f). Lalas *et al* (2007) discuss potential management actions pertaining to this threat. When the presence of the sea lion at two intensively managed sites is considered, the positive effect of intensive management on growth rate appears even stronger. After accounting for the sea lion, intensive management was responsible for a 0.0847 average increase in log growth rate, or an 8.8% average increase in growth rate; see Table 3(f). Intensive management was responsible for 53 additional yellow-eyed penguin nests by 2006—a 13% increase on the counterfactual. This is equivalent to 0.93 average additional nests per site-year of treatment over all intensively managed site-years, or NZ\$49,500 per additional nest; see Table 8(b). While we suspect that these figures come closer to estimating the true effect of intensive management, we choose to highlight the

more conservative figures that do not account for the sea lion to avoid cherry picking included variables.

Notably, this analysis finds that yellow-eyed penguin population growth rate is negatively correlated with nest density; see Table 3(b-h). While a density dependent growth rate is expected by ecological theory, previous work did not find evidence of density dependence (McKinlay, 1997; Alexander and Shields, 2003). Density dependence could imply decreasing returns to conservation activities as carrying capacity is approached at a site.

Following the inclusion of year dummies, the coefficient of determination increases from 0.05 to 0.19 (Table 3(b-c)), suggesting that good years and bad years for penguins are broadly correlated across sites. Food availability at sea has long been established as a contributing factor to good and bad yellow-eyed penguin breeding years (Richdale, 1957; Darby and Seddon, 1990; van Heezik and Davis, 1990). This suggests the potential for yellow-eyed penguins to be aided by marine-based management interventions. Intertemporal variation in growth rate has also been caused by avian malaria (Graczik *et al*, 1995), toxins (Gill and Darby, 1993), and La Niña events (Moore and Wakelin, 1997). These may contribute more than terrestrial factors to penguin mortality.

Discussion

The ideal way to test the effectiveness of a conservation treatment is with a before-after, control-impact pairs (BACIP) study (Stewart-Oaten *et al*, 1986). However,

this requires pre-project planning. Conservationists are frequently faced with the challenge of analyzing the impact of treatments without the benefit of proactive study design. With a large enough panel data set, econometric techniques can be used to perform after-the-fact analysis of the impact of treatments. In this paper we have performed this type of analysis for three yellow-eyed penguin recovery treatments.

In this study we find that intensive management is positively correlated with increases in annual site-level yellow-eyed penguin population growth rate, while trapping of predators and revegetation are not. Our findings are consistent with two yellow-eyed penguin population viability analyses, each showing that growth rate is most sensitive to changes in adult mortality. Our findings are also consistent with a necropsy study that found that more than half of yellow-eyed penguin specimens collected died of the stresses that intensive management is intended to reduce.

We estimate that intensive management raised the number of nests in 2006 by 9% from the counterfactual, from 424 to 462. The intensive management that provided these 38 additional nests would have cost DOC an estimated NZ\$2.6 million in labor and materials, or NZ\$68,000 per additional nest. Where volunteer labor is available, or where intensive management can be added to the duties of already paid staff, additional nests could be provided more cheaply. The cost per additional nest should be considered in relation to non-market values these penguins provide (Schanzel and McIntosh, 2000), as well as the estimated several million dollars in revenue generated annually by penguin tourism (Tisdell, 2007), much of which is concentrated at a few sites.

In light of our findings, conservation groups with access to sufficient funding or volunteer labor might consider expanding intensive management to new penguin sites.

Not only is intensive management the most promising of the three terrestrial recovery treatments analyzed, but expanding intensive management to new sites would provide additional site level before-and-after data for identification in a site fixed effects model.

Expanded intensive management could be combined with controlled ecological testing of specific intensive management mechanisms to determine which are responsible for increased population growth rates. If periodic maintenance of traps and nest boxes is contributing to growth rates, these activities could be expanded at relatively low cost to other managed sites. If translocation is found to be contributing to the success of intensively managed sites, managers may consider intensive management as a method for restoring yellow-eyed penguins to sites which they no longer currently occupy. Trapping and revegetation could also be subjected to controlled ecological testing, either across or within sites.

The methods employed here for evaluating effectiveness of yellow-eyed penguin recovery treatments can be extended to any conservation program anywhere, though a few key elements contributed to the evaluation in this case. Yellow-eyed penguin populations live at many sites, and are stationary from year to year, with little movement of individuals between sites. A comprehensive data set of a robust population metric, in this case nests, were counted at every site using consistent methodology.

Unless the species is in critical danger, managers should consider leaving some sites as controls, deliberately withholding treatment. Control sites are critical to determining the counterfactual—what would have happened to a population if no treatment had been applied. This evaluation benefited from a multitude of control sites, as well as from a diversity of treatments employed across sites. Analysis is easiest when

treatment and control sites are randomly selected, though as we have shown, panel econometrics can at least partially compensate for non-random site selection.

When after-the-fact analysis is performed, analysts must confront the possibility that treatments may not have been randomly assigned across sites. Locations for treatments may instead have been targeted to sites where treatments were expected to have the greatest impact, or to sites which were most easily accessible. In this paper we have demonstrated three ways to confront non-random assignment of treatments. First, control variables for site characteristics can be included in the regression. Second, a site fixed effects model can account for intertemporal variation in treatment at a site. And finally, a two-stage least squares model can be applied, using site accessibility as an instrumental variable. For this to be effective, site accessibility must be plausibly correlated with the probability of treatment but not correlated with population growth rate. This instrument is only useful for species such as the yellow-eyed penguin, for which human contact does not have a major influence on growth rate. Selection bias may never be completely eliminated with these techniques, which strengthens the case for controlled ecological experiments or the random assignment of treatments across available sites.

A population monitoring program and a time-variant mixture of treatment and control sites allows for an after-the-fact econometric analysis of which conservation treatments are most effective. A disaggregated record of the expenditures associated with each treatment allows analysis of which conservation treatments provide best value for money as well. Even an ineffective or costly program can be valuable if we can learn from its shortcomings. On the other hand, a treatment which has been shown to be

effective or cost effective should attract further resources for investment. We submit the case of the yellow-eyed penguin in the hope that effectiveness and cost effectiveness analysis will be extended to conservation in other settings.

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References

Abbitt, R. and J. Scott (2001). Examining differences between recovered and declining endangered species. *Conservation Biology*, 15:1274-1284.

Alexander, R.R. and D.W. Shields (2003). Using land as a control variable in density - dependent bioeconomic models. *Ecological Modelling*, 170:193-201.

Birdlife International (2007a). *Megadyptes antipodes*. In: IUCN 2007. 2007 IUCN Red List of Threatened Species. <www.iucnredlist.org>.

Birdlife International (2007b). *Spheniscus mendiculus*. In: IUCN 2007. 2007 IUCN Red List of Threatened Species. <www.iucnredlist.org>.

Birdlife International (2007c). *Eudyptes sclateri*. In: IUCN 2007. 2007 IUCN Red List of Threatened Species. <www.iucnredlist.org>.

Bruner, A.G., R.E. Gullison, R.E. Rice, and G.A.B. da Fonseca (2001). Effectiveness of parks in protecting tropical biodiversity. *Science*, 5:125-128.

Buchan, D. (2007). Not just trees in the ground: The social and economic benefits of community-led conservation projects. WWF-New Zealand, Wellington.

Cullen, R., G. A. Fairburn, and K. F. D. Hughey (2001). Measuring the productivity of threatened species programs. *Ecological Economics*, 39: 53–66.

Cullen, R., E. Moran, and K. F. D. Hughey (2005). Measuring the success and cost effectiveness of New Zealand multiple species programs. *Ecological Economics*, 53:311-323.

Darby, J.T. (1996). Presentation. 3rd International Penguin Conference. Cape Town, South Africa.

Darby, J.T. and S.M. Dawson (2000). Bycatch of yellow-eyed penguins (*Megadyptes antipodes*) in gillnets in New Zealand waters 1979–1997. *Biological Conservation*, 93:327–332.

Darby, J.T. and P.J. Seddon (1990). Breeding biology of yellow-eyed penguins (*Megadyptes antipodes*). Pp. 45-62 in: Davis, L.S.; Darby, J.T. (ed.) *Penguin Biology*. Academic Press, New York, USA.

Department of Conservation (2004). Annual report for the year ended 30 June 2004.

Efford, M.G. and K.A. Edge (1998). Can artificial brood reduction assist the conservation of yellow-eyed penguins (*Megadyptes antipodes*)? *Animal Conservation*, 1:263-271.

Efford, M., N. Spencer, and J. Darby (2004). Population studies of yellow-eyed penguins, 1993-94 progress report. Landcare Research contract report LC9495/39, prepared for Department of Conservation.

Ellenberg, U., A.N. Setiawan, A. Cree, D.M. Houston, P.J. Seddon (2007). Elevated hormonal stress response and reduced reproductive output in yellow-eyed penguins exposed to unregulated tourism. *General and Comparative Endocrinology*, 152:54-63.

Engeman, R.M., S.A. Shwiff, B. Constantin, M. Stahl, and H.T. Smith (2002). An economic analysis of predator removal approaches for protecting marine turtle nests at Hobe Sound National Wildlife Refuge. *Ecological Economics*, 42:469-478.

Ferraro, P.J., C. McIntosh, and M. Ospina (2007). The effectiveness of listing under the U.S. Endangered Species Act: An econometric analysis using matching methods. *Journal of Environmental Economics and Management*, 54(3):245-261.

Ferraro, PJ and SK Pattanayak (2006) Money for Nothing? A Call for Empirical Evaluation of Biodiversity Conservation Investments. *PloS Biology*, 4(4): p0482-0488.

Gill, J.M. and J.T. Darby (1993). Deaths in Yellow-Eyed Penguins (*Megadyptes antipodes*) on the Otago Peninsula during the summer of 1990. *New Zealand Vet Journal*, 41:39-42.

Graczyk, T.K., J.F. Cockrem, M.R. Cranfield, J.T. Darby, and P. Moore (1995). Avian malaria seroprevalence in wild New Zealand penguins. *Parasite*, 2:401-405.

Halpern, B.S., C.R. Pyke, H.E. Fox, C. Haney, M.A. Schlaepfer, and P. Zaradic (2006). Gaps and mismatches between global conservation priorities and spending. *Conservation Biology*, 20(1):56-64.

Hitchmough, R., L. Bull, and P. Cromarty (2005). New Zealand threat classification system lists: 2005. Science & Technical Publishing, Department of Conservation, Wellington, New Zealand.

Hocken, A.G. (2005). Necropsy findings in yellow-eyed penguins (*Megadyptes antipodes*) from Otago, New Zealand. *New Zealand Journal of Zoology*, 32 :1-8.

Innes, J., R. Hay, I. Flux, P. Bradfield, H. Speed, and P. Jansen (1999). Successful recovery of North Island kokako *Callaeas cinerea wilsoni* populations, by adaptive management. *Biological Conservation*, 87(2):201-214.

IUCN (2007). The IUCN red list of threatened species. Table 5 : Threatened species in each country (totals by taxonomic group). Gland, Switzerland.

James, AN, KJ Gaston, and A. Balmford (1999). Balancing the Earth's accounts. *Nature*, 401:323-324.

Kennedy, E. (2007). Model or muddle? Does the charitable trust concept serve species conservation in New Zealand? *Conserv-Vision: The next 50 years*. Hamilton, New Zealand.

Kiesecker, J.M., T. Comendant, T. Grandmason, E. Gray, C. Hall, R. Hilsenbeck, P. Kareiva, L. Lozier, P. Naehu, A. Rissman, M.R. Shaw, and M. Zankel (2007). Conservation easements in context: A quantitative analysis of their use by The Nature Conservancy. *Frontiers in Ecology and the Environment*, 5:125-130.

Lalas, C., H. Ratz, K. McEwan, and S.D. McConkey (2007). Predation by New Zealand sea lions (*Phocarctos hookeri*) as a threat to the viability of yellow-eyed penguins (*Megadyptes antipodes*) at Otago Peninsula, New Zealand. *Biological Conservation*, 135:235-246.

Lotka, A. J. (1925). *Elements of Physical Biology*. Williams and Wilkins, pub., Baltimore.

MapToaster Topo/NZ (2007). *Integrated Mapping*, Christchurch, New Zealand.

Marchant, S. and P.J. Higgins (1990). *The Handbook of Australian, New Zealand and Antarctic Birds*. Volume I: Ratites to Ducks. Oxford University Press, Melbourne, Australia.

McClung, M.R., P.J. Seddon, M. Massaro, A.N. Setiawan (2003). Nature-based tourism impacts on yellow-eyed penguins *Megadyptes antipodes*: Does unregulated visitor access affect fledging weight and juvenile survival? *Biological Conservation*, 119:279-285.

McKinlay, B. (1997). The conservation of yellow-eyed penguins (*Megadyptes antipodes*): Use of a PVA model to guide policy development for future conservation management direction. University of Otago Wildlife Management Report Number 97. Dunedin, New Zealand.

McKinlay, B. (2001). Hoiho (*Megadyptes antipodes*) recovery plan, 2000-2025. Threatened species recovery plan 35. Department of Conservation, Wellington, New Zealand.

Millennium Ecosystem Assessment (MEA) (2005). Ecosystems and human well-being: Policy Responses: Findings of the Responses Working Group of the Millennium Ecosystem Assessment. Island Press, Washington, D.C.

Ministry of Tourism (2007). Activities/Attractions in NZ. International Visitor Survey: YE March 2007.

Ministry of Tourism (2007). Average visitor spend by purpose. International Visitor Survey: YE March 2007.

Moore, P.J. (1999). Foraging range of the yellow-eyed penguin (*Megadyptes antipodes*). *Marine Ornithology*, 27:49-58.

Moore, P.J. (2001). Historical records of yellow-eyed penguin (*Megadyptes antipodes*) in southern New Zealand. *Notornis*, 48:145-156.

Moore, P.J. and M.D. Wakelin (1997). Diet of Yellow-Eyed Penguin (*Megadyptes antipodes*) on Campbell and Auckland Islands 1987 -1990. *Notornis*, 39:1-15.

Moorhouse, R., T. Greene, P. Dilks, R. Powlesland, L. Moran, G. Taylor, A. Jones, J. Kneegmans, D. Wills, M. Pryde, I. Fraser, A. August, and C. August. Control of introduced mammalian predators improves kaka *Nestor meridionalis* breeding success: Reversin the decline of a threatened New Zealand parrot. *Biological Conservation*, 110:33-44.

O'Donnell, C.F.J. (1996). Predators and the decline of New Zealand forest birds: An introduction to the hole-nesting bird and predator programme. *New Zealand Journal of Zoology*, 23:213-219.

O'Donnell, C.F.J., P.J. Dilks, and G.P. Elliott (1996). Control of a stoat (*Mustela erminea*) population irruption to enhance mohua (yellowhead) (*Mohoua ochrocephala*) breeding success in New Zealand. *New Zealand Journal of Zoology*, 23:279 -286.

Ratz, H (1997). Ecology, identification, and control of introduced mammalian predators of yellow-eyed penguin. Unpublished PhD thesis, University of Otago, Dunedin, New Zealand.

Richdale, L.E. (1957). A population study of penguins. Clarendon Press, Oxford, UK.

Sanchez-Azofeifa, G.A., A. Pfaff, J.A. Robalino, and J.P. Boomhower (2007). Costa Rica's payment for environmental services program: Intention, implementation, and impact. *Conservation Biology*, 21(5):1165-1173.

Schanzel, H. and A.J. McIntosh (2000). An insight into the personal and emotive context of wildlife viewing at the Penguin Place, Otago Peninsula, New Zealand. *Journal of Sustainable Tourism*, 8(1): 36-52.

Seddon, P.J. and L.S. Davis (1989). Nest site selection by yellow-eyed penguins. *Condor*, 91: 653-659.

Seddon, P.J., van Heezik, Y.M., Darby, J.T. (1989). Inventory of yellow-eyed penguin (*Megadyptes antipodes*) mainland breeding areas, South Island, New Zealand.

Unpublished report commissioned by the Yellow-eyed Penguin Trust and the Otago Branch of the Royal Forest and Bird Protection Society of New Zealand Inc. 182 pp.

Seddon, P.J. (1990). Behaviour of the yellow-eyed penguin chick. *Journal of Zoology* (London), 220:333-343.

Shumway, S.E., S.M. Allen, and P.D. Boersma (2003). Marine birds and harmful algal blooms: Sporadic victims or under-reported events? *Harmful Algae*, 2:1-17.

Shwiff, S., R.T. Sterner, J. W. Turman, B.D. Foster (2005). Ex post economic analysis of reproduction-monitoring and predator-removal variables associated with protection of the endangered California least tern. *Ecological Economics*, 53:277-287.

Sills, E., R. Arriagada, P. Ferraro, S. Pattanayak, L. Carrasco, E. Ortiz, S. Cordero, and K. Andam. Forthcoming. Impact of the PSA Program on Land Use. In *Ecomarkets: Costa Rica's Experience with Payments for Environmental Services*, edited by G. Platais and S. Pagiola.

Stewart-Oaten, A., W.W. Murdoch, and K.R. Parker (1986). Environmental impact assessment: "Pseudoreplication" in time? *Ecology*, 67(4):929-940.

Taylor, G.A. (2000). Action plan for seabird conservation in New Zealand, Part A: Threatened seabirds. Threatened species occasional publication 16. New Zealand Department of Conservation, Wellington, New Zealand, 234pp.

Terralink (2007). New Zealand Land Cover Data base.

Tisdell, C. (2007). The economic importance of wildlife conservation on the Otago Peninsula – 20 years on. Economics, Ecology and the Environment Working Paper No. 144, University of Queensland, Brisbane, Australia.

Van Heezik, Y and L. Davis (1990). Effects of food variability on growth rates, fledging sizes, and reproductive success in the Yellow-Eyed Penguin (*Megadyptes antipodes*). Ibis, 132:354-365.

Wilson, K.A, E.C. Underwood, S.A. Morrison, K.R. Klausmeyer, W.M Murdoch, B. Reyers, G. Wardell-Johnson, P.A. Marquet, P.W. Rundel, M.F. McBride, R.L. Pressey, M. Bode, J.M. Hoekstra, S. Andelman, M. Looker, C. Rondinini, P. Kareiva, M.R. Shaw, and H.P. Possingham (2007). Conserving biodiversity efficiently: What to do, where, and when. PloS Biology, 5(9).

	Overall		Trapped		Revegetated		Intensively Managed		No treatment	
Total Site-Years	720		204		155		57		448	
Total Observations ($n_{it} \geq 0$)	646		197		149		57		383	
Positive Observation ($n_{it} > 0$)	540		176		118		52		301	
Jsable Observations ($n_{it}, n_{it-1} > 0$)	519		172		115		52		285	
Region										
North Otago	105		23		12		17		76	
Otago Peninsula	229		88		58		35		96	
Catlins	185		61		45		0		113	
	Mean	St. Dev	Mean	St. Dev	Mean	St. Dev	Mean	St. Dev	Mean	St. Dev
Nests	12.3	10.2	16.3	10.5	13.3	9.5	19.0	10.2	9.4	9.0
Log lambda	0.0025	0.18	0.0071	0.17	-0.0052	0.17	0.0209	0.17	0.0039	0.18
Lambda	1.10	0.54	1.10	0.51	1.06	0.42	1.14	0.56	1.11	0.58
Area (Ha)	7.3	5.8	8.6	5.8	6.2	4.2	5.65	3.96	6.9	6.0
Population density (Nests/Ha)	2.4	2.2	2.7	2.6	2.8	1.8	4.8	3.9	2.0	1.8
Land Cover (%)										
Grassland	64.3	32.9	76.6	26.2	71.2	24.5	77.8	23.9	55.9	34.9
Shrubland	4.9	13.2	3.3	11.5	7.4	19.0	0	0	5.3	12.4
Forest	12.1	24.4	8.6	18.7	6.8	15.2	0	0	16.2	28.6
Sand and gravel	16.6	20.2	11.6	15.7	14.6	11.6	22.2	23.9	18.9	21.6
Other	2.1	7.7	0	0	0	0	0	0	3.8	10.0
Mean elevation (m)	41.2	46.9	42.0	33.1	31.7	22.8	18.8	14.7	42.9	57.2
Mean slope (%)	12.7	8.5	13.9	9.5	10.2	6.5	8.2	4.9	12.4	8.4
Distance from Dunedin (km)	76.4	56.1	64.7	54.3	75.7	58.9	45.3	21.1	88.3	55.7
Distance from road (m)	862.6	615.5	928.9	432.8	919.2	600.8	942.3	257.1	796.9	702.0
Initial Population ¹ (Nests)	10.3	8.4	13.5	8.9	12.3	8.1	16.4	10.3	7.8	7.3
Initial Density ¹ (Nests/Ha)	1.9	1.6	1.9	1.3	2.7	1.7	2.8	1.6	1.6	1.5

Table 1 – Summary statistics

¹In 1992 (37 sites) or year of earliest nest counts (11 sites)

Revegetation	First Year	Years 2, 4, 6 only
Materials	\$3 per plant 2500 plants per hectare	
Labor	\$16 per person-hour 225 person-hours	\$16 per person hour 36 person-hours
Total	\$11,100.00 per hectare	\$576.00 per hectare
Trapping	First Year	Each Subsequent Year
Materials	\$50 per new trap 5 traps per hectare	\$50 per replacement trap 0.5 traps per hectare
	\$12.50 bait and poison per trap 5 traps per hectare	\$12.50 bait and poison per trap 5 traps per hectare
Labor	\$16 per person-hour 52 person-hours per hectare	\$16 per person-hour 52 person-hours per hectare
Total without intensive management (Materials plus labor)	\$1,144.50 per hectare	\$919.50 per hectare
Total with intensive management (Materials only)	\$312.50 per hectare	\$87.50 per hectare
Intensive Management	First Year	Each Subsequent Year
Materials	\$50 food and medicine per nest	\$50 food and medicine per nest
Labor	\$40,000 full time ranger per site	\$40,000 full time ranger per site
Total	\$40,000 per site +\$50 per nest	\$40,000 per site +\$50 per nest

Table 2 – Indicative average cost of treatments per hectare (2007 NZ\$)

	(a)	(b)	(c)	(d)	(e)	(f)	(g)	(h)	(i)
Observations	519	506	506	506	506	506	506	506	506
Intercept	0.0007 (0.07)	0.02013* (1.89)	0.0478 (1.51)	0.0516* (1.65)	0.0475 (1.50)	0.0837 (1.43)	0.0471 (1.48)	0.0485 (1.30)	0.1476*** (3.91)
Trapping	0.0091 (0.50)	0.0096 (0.54)	0.0151 (0.89)	-0.0055 (-0.31)	0.0116 (0.68)	0.0134 (0.68)	0.0027 (0.13)	0.0225 (1.15)	0.0013 (0.04)
Revegetation	-0.0140 (-0.69)	-0.0041 (-0.21)	-0.0107 (-0.57)	0.0164 (0.82)	-0.0050 (-0.26)	-0.0061 (-0.30)	-0.0039 (-0.14)	-0.0135 (-0.65)	-0.0047 (-0.12)
Intensive management	0.0185 (0.69)	0.0582** (2.13)	0.0523** (2.03)	0.0478* (1.72)	0.0847*** (2.64)	0.0828** (2.46)	0.0361 (0.76)	0.0741** (2.22)	0.0807 (0.79)
Log density	no	-0.1104*** (-5.12)	-0.0998*** (-4.82)	-0.1002*** (-4.84)	-0.1009*** (-4.89)	-0.1211*** (-5.25)	-0.1022*** (-4.86)	-0.1055*** (-4.99)	-0.5376*** (-12.93)
Year dummies	no	no	yes	yes	yes	yes	yes	yes	yes
Three year lag	no	no	no	yes	no	no	no	no	no
Sea lion	-	-	-	-	-0.0813* (-1.69)	-0.0867* (-1.68)	-0.0335 (-0.567)	-	-0.0944 (-1.46)
Fire	-	-	-	-	-	-0.4391** (-2.59)	-	-	-
North Otago Region	-	-	-	-	-	(dropped)	-	-	-
Otago Peninsula Region	-	-	-	-	-	-0.0057 (-0.17)	-	-	-
Catlins Region	-	-	-	-	-	-0.0100 (-0.28)	-	-	-
Grassland (%)	-	-	-	-	-	-0.0155 (-0.351)	-	-	-
Shrubland (%)	-	-	-	-	-	-0.0176 (0.22)	-	-	-
Forest (%)	-	-	-	-	-	-0.0245 (-0.47)	-	-	-
Mean elevation (m)	-	-	-	-	-	-0.0002 (-0.95)	-	-	-
Mean slope (%)	-	-	-	-	-	0.0002 (0.17)	-	-	-
Distance from Dunedin (km)	-	-	-	-	-	-1.00x10 ⁻⁴ (-0.25)	-	-	-
Distance from road (m)	-	-	-	-	-	7.43x10 ⁻⁷ (-0.05)	-	-	-
Trapping x revegetation	-	-	-	-	-	-	0.0083 (0.20)	-	-
Trapping x intensive management	-	-	-	-	-	-	0.0983 (1.54)	-	-
Revegetation x intensive management	-	-	-	-	-	-	-0.0977 (-1.24)	-	-
Trapping within 10 km	-	-	-	-	-	-	-	-0.0052 (-1.01)	-
Revegetation within 10 km	-	-	-	-	-	-	-	0.0027 (0.28)	-
Intensive management within 10km	-	-	-	-	-	-	-	0.0082 (0.77)	-
Site fixed effects	no	no	no	no	no	no	no	no	yes
R ²	0.002	0.05	0.19	0.19	0.19	0.20	0.20	0.20	#
Adjusted R ²	-0.004	0.04	0.16	0.16	0.17	0.16	0.16	0.16	#

Table 3 – Effect of treatments on site-year log growth rate

OLS regression; t-statistic in parentheses

*Significant at $p < 0.10$

**Significant at $p < 0.05$

***Significant at $p < 0.01$

Overall R²=0.09; Within R²=0.39; Between R²=0.04.

Increment	1	2	3	4	5	6	7
Observations	506	466	425	385	345	301	259
Intercept	0.0478 (1.51)	0.2420*** (3.04)	0.1406* (1.66)	0.1780* (1.88)	0.4479*** (4.74)	0.0180 (0.17)	0.0975 (0.85)
Trapping	0.0151 (0.89)	0.0133 (0.57)	0.0061 (0.35)	-0.0018 (-0.11)	-0.0080 (-0.59)	-0.0090 (-0.70)	-0.0055 (-0.43)
Revegetation	-0.0107 (-0.57)	0.0022 (0.09)	-0.0001 (-0.01)	-0.0005 (-0.03)	0.0006 (0.04)	0.0035 (0.25)	0.0076 (0.56)
Intensive Management	0.0523** (2.03)	0.0929*** (2.68)	0.0821*** (3.17)	0.0803*** (3.43)	0.0768*** (3.76)	0.0735*** (3.83)	0.0715*** (3.81)
Log Density	-0.0998*** (-4.82)	-0.1274*** (-5.32)	-0.1498*** (-5.60)	-0.2036*** (-6.38)	-0.2023*** (-5.80)	-0.2395*** (-6.11)	-0.2915*** (-6.63)
Year dummies	yes	yes	yes	yes	yes	yes	yes
R ²	0.19	0.28	0.34	0.31	0.28	0.31	0.33

Table 4 – Effect of treatments on site -year log growth rate (multiple year sub -samples)

OLS regression; t-statistic in parentheses

*Significant at $p < 0.10$

**Significant at $p < 0.05$

***Significant at $p < 0.01$

Increment	8	9	10	11	12	13	14
Observations	222	185	150	117	86	56	28
Intercept	-0.0924 (-0.75)	-0.0212 (0.15)	-0.0828 (-0.56)	0.1654 (1.10)	0.1832 (1.06)	0.08870 (0.55)	0.0844 (0.37)
Trapping	-0.0046 (-0.36)	0.0001 (0.01)	0.0018 (0.13)	0.0100 (0.64)	0.0123 (0.68)	0.0125 (0.64)	0.0232 (0.77)
Revegetation	0.0104 (0.76)	0.0119 (0.84)	0.0131 (0.90)	0.0113 (0.71)	0.0163 (0.89)	0.0226 (1.14)	0.0228 (0.72)
Intensive Management	0.0694*** (3.67)	0.0644*** (3.32)	0.0602*** (3.04)	0.0509** (2.41)	0.0482** (2.05)	0.0503** (2.01)	0.0455 (1.19)
Log Density	-0.3270*** (-6.46)	-0.3712*** (-6.30)	-0.3818*** (-5.55)	-0.4085*** (-5.17)	-0.4605*** (-4.69)	-0.4285*** (-3.81)	-0.5462** (-2.68)
Year dummies	yes	yes	yes	yes	yes	yes	n.a.
R ²	0.32	0.29	0.22	0.22	0.26	0.27	0.28

Table 4 (cont'd.) – Effect of treatments on site-year log growth rate (multiple year sub -samples)

OLS regression; t-statistic in parentheses

*Significant at $p < 0.10$

**Significant at $p < 0.05$

***Significant at $p < 0.01$

Dependent Variable	Trapping	Revegetation	Intensive Management
Observations	526	526	526
Intercept	0.3233*** (3.45)	0.3526*** (4.18)	0.1642*** (2.85)
Instruments			
Distance from Dunedin (km)	-0.0008** (-2.06)	0.0004 (1.20)	-0.0005** (-2.38)
Mean slope (%)	0.0034 (1.40)	-0.0086*** (-3.85)	-0.0065*** (-4.24)
Initial population ¹ (nests)	0.0116*** (4.35)	0.0073*** (3.04)	0.0065*** (3.98)
Log density (nests/Ha)	-0.027 (0.45)	0.0470 (0.87)	0.1399*** (3.78)
Year dummies	yes	yes	yes
F-statistic on instruments	11.80	6.58	11.30
R ²	0.10	0.07	0.15

Table 5 – Two stage least squares regression: First stage results

OLS, t-statistic in parentheses

¹In year 1992 (37 sites) or year of earliest nest counts (11 sites)

*Significant at $p < 0.10$

**Significant at $p < 0.05$

***Significant at $p < 0.01$

Observations	506
Intercept	0.0794 (0.99)
Trapping	-0.0427 (-0.48)
Revegetation	-0.0924 (-0.51)
Intensive Management	0.2281 (1.20)
Log density	-0.1360*** (-3.48)
Year dummies	yes
Used as Instruments:	
-Distance from Dunedin (km)	
-Mean slope (%)	
-Initial Population ¹ (nests)	
Root MSE	0.18
Hausman (f-statistic on residuals)	0.59
Hausman (p value)	0.62

Table 6 – Two stage least squares regression: Second stage results

Two-stage least squares (2SLS); t-statistic in parentheses

¹In year 1992 (37 sites) or year of earliest nest counts (11 sites)

*Significant at $p < 0.10$

**Significant at $p < 0.05$

***Significant at $p < 0.01$

	(a)			(b)		
Prediction Model	3©—Without Sea Lion			3(f)—Including Sea Lion		
Actual nests, N_{2006}	462.5			462.4		
Coefficient of treatment magnitude	$\hat{b}_{.025}$	$\hat{b}_{.5}$	$\hat{b}_{.975}$	$\hat{b}_{.025}$	$\hat{b}_{.5}$	$\hat{b}_{.975}$
Trapping						
Counterfactual nests, \hat{N}_{2006}	523.5	421.1	352.2	537.5	429.9	357.8
Nests gained from treatment (total)	-61.0	41.4	110.3	-75.1	32.5	104.6
Site-years receiving treatment		204			204	
Nests gained from treatment per site-year	-0.30	0.20	0.54	-0.37	0.16	0.51
Total replacement cost of treatment (NZ\$)		\$1,456,667			\$1,456,667	
Average cost of treatment per site-year (NZ\$)		\$7,141			\$7,141	
Nests gained from treatment per NZ\$100,000	-4.19	2.84	7.57	-5.16	2.23	7.18
Average cost per additional nest (NZ\$)	Inf.	\$35,188	\$13,207	Inf.	\$44,823	\$13,927
Revegetation						
Counterfactual nests, \hat{N}_{2006}	559.7	480.3	425.8	546.1	470.5	418.6
Nests gained from treatment (total)	-97.2	-17.8	36.7	-83.7	-8.1	43.8
Site-years receiving treatment		140			140	
Nests gained from treatment per site-year	-0.69	-0.13	0.26	-0.60	-0.06	0.31
Total replacement cost of treatment (NZ\$)		\$1,344,252			\$1,344,252	
Average cost of treatment per site-year (NZ\$)		\$9,602			\$9,602	
Nests gained from treatment per NZ\$100,000	-7.23	-1.32	2.73	-6.23	-0.60	3.26
Average cost per additional nest (NZ\$)	Inf.	Inf.	\$36,628	Inf.	Inf.	\$30,691
Intensive Management						
Counterfactual nests, \hat{N}_{2006}	461.0	424.3	403.4	443.8	409.5	392.0
Nests gained from treatment (total)	1.5	38.2	59.1	18.6	52.9	70.4
Site-years receiving treatment		57			57	
Nests gained from treatment per site-year	0.03	0.67	1.04	0.33	0.93	1.24
Total replacement cost of treatment (NZ\$)		\$2,619,350			\$2,619,350	
Average cost of treatment per site-year (NZ\$)		\$45,954			\$45,954	
Nests gained from treatment per NZ\$100,000	0.06	1.46	2.26	0.71	2.02	2.69
Average cost per additional nest (NZ\$)	\$1,746,233	\$68,569	\$44,321	\$140,827	\$49,516	\$37,207

Table 7 – Average cost of providing an additional nest using each treatment

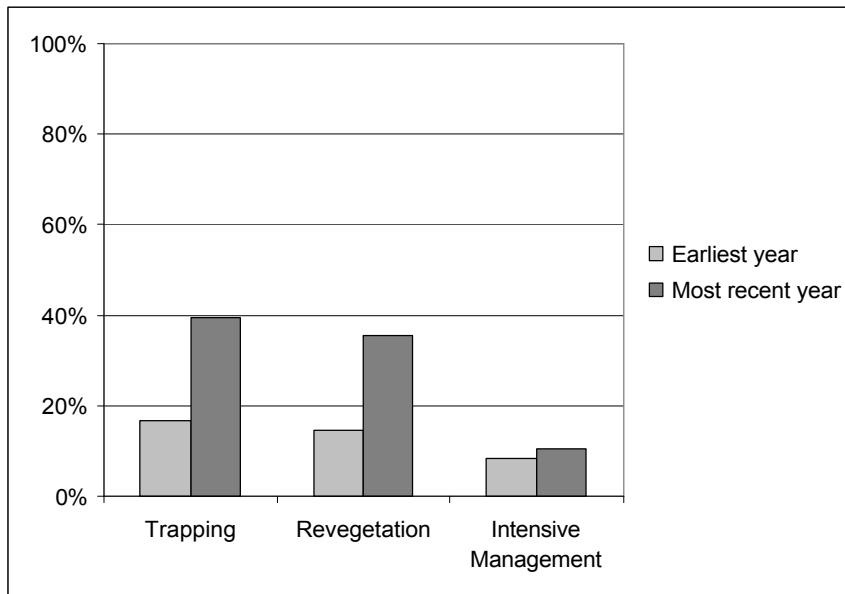


Figure 1 – Percent of sites receiving each treatment during earliest year¹ and most recent year²

¹Earliest year=1992 at 37 sites ; >1992 at 11 sites

²Most recent year=2006 at 40 sites; <2006 at 8 sites

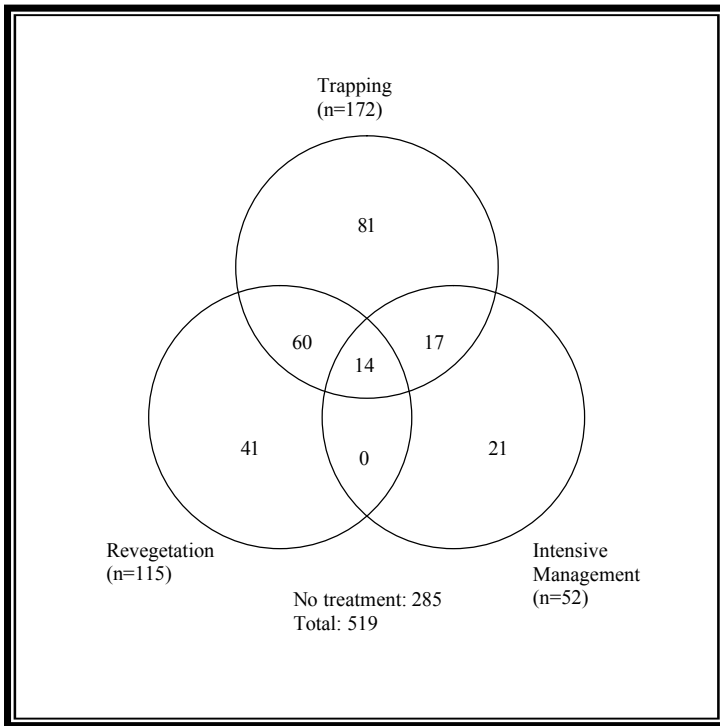


Figure 2 – Number of site -years receiving each combination of treatments

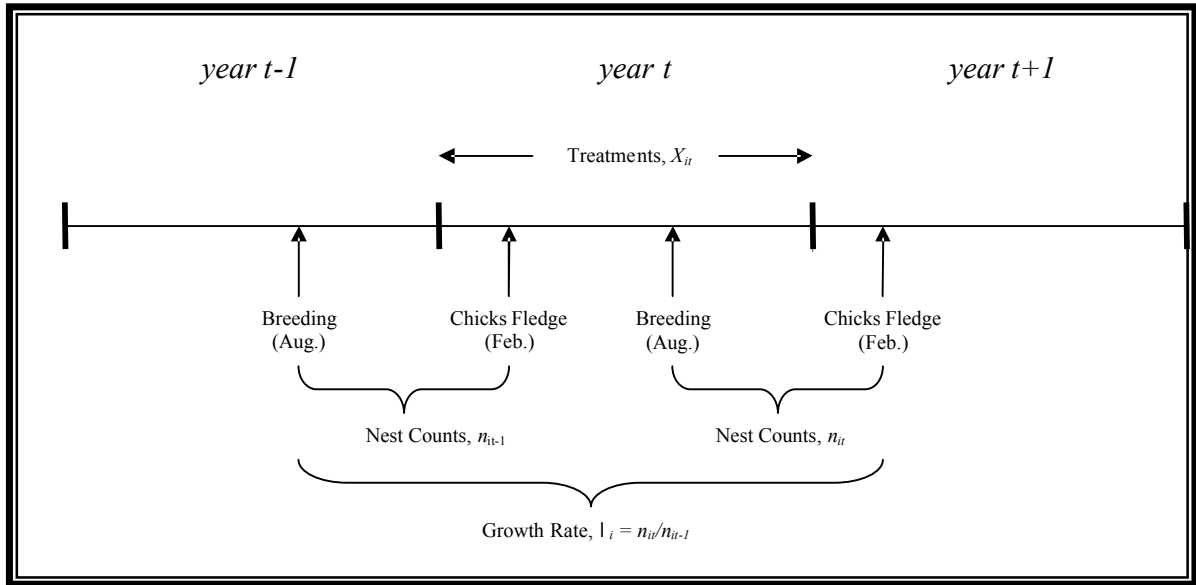


Figure 3 – Timeline of treatments and nest counts