

AARES 52nd Annual Conference

2008

Effectiveness and cost effectiveness of Yellow-Eyed Penguin conservation measures

Jonah Busch and Ross Cullen

Effectiveness and cost effectiveness of Yellow-Eyed Penguin conservation measures

Jonah Busch¹ and Ross Cullen²

¹ (Author to whom correspondence should be addressed)

Donald Bren School of Environmental Science and Management
4520 Bren Hall, University of California, Santa Barbara, CA, 93106, USA.
busch@umail.ucsb.edu

²Professor of Resource Economics, Commerce Division
Commerce 202, Lincoln University, Lincoln, NEW ZEALAND
Working paper—Comments welcome; please do not cite

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. Conserving biodiversity efficiently depends on identifying conservation treatments which provide greatest return on investment. Where controlled experiments are not possible, panel econometric techniques can be used to determine the effectiveness of conservation treatments. A long-running Yellow-Eyed Penguin (*Megadyptes antipodes*) nest count in New Zealand presents a golden opportunity to compare the effectiveness and cost effectiveness of three commonly used conservation 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. We confront the possibility of selection bias in treatment with site fixed effects and with an instrumental variable based on site accessibility. Of the three treatments analyzed, only intensive management is significantly correlated with increases in site-level penguin population growth rate. We estimate the marginal cost of providing yellow-eyed penguins through intensive management to be NZ\$68,600 per nest.*

Introduction

Conserving our planet's biological diversity is a grand investment. An estimated US\$6 billion is spent annually on nature reserves worldwide (James *et al*, 1999). International conservation organizations spent \$1.5 billion in 2002 (Halpern *et al*, 2006). NZ\$106.5 million was spent on management of natural heritage in New Zealand (DOC, 2004). Despite the magnitude of this financial outlay, little research has been conducted on which investments are most successful. The Millenium 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.0482) 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 will enable conservationists to direct scarce resources to the most effective treatments (Wilson *et al*, 2007). Additionally, resolving uncertainty associated

with the cost effectiveness of a conservation approach may attract more funding for conservation.

The effect of a conservation treatment can be tested through a controlled experiment such as a before-after, control-impact, paired (BACIP) study (Stewart-Oaten *et al*, 1986). However, such a study must be carefully planned in advance of providing conservation treatment; BACIP can't be performed after a treatment regimen is already in place. Furthermore, conservation treatments are often applied in a deliberately non-random fashion, for instance to those sites where treatment is expected to be most successful. When a BACIP study is infeasible, panel econometric techniques can be used to test effects of conservation treatment. This paper uses a long-running restoration program for the endangered yellow-eyed penguin to demonstrate how panel econometrics can be used to evaluate the effectiveness of three commonly used species restoration techniques.

A small but growing body of economic literature evaluates the effectiveness of conservation programs. Bruner *et al* (2001) found that land within tropical protected areas suffered less forest cover loss than adjacent land outside protected areas. Ferraro *et al* (2007) found that species listed under the U.S. Endangered Species Act only showed recovery relative to comparable non-listed species if their listing was accompanied by funding. Sills *et al* (forthcoming) examine the effectiveness of Costa Rica's Payments for Ecosystem Services program in deterring deforestation.

Some research has gone a step further, calculating cost effectiveness of conservation programs, that is, the improvement in biological outcome obtained 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) compares 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.

Efforts to determine success 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). This makes instances in which good records have been kept of management actions and species performance especially valuable for evaluating conservation performance. A long-running recovery program for the yellow-eyed penguin (*Megadyptes antipodes*) in New Zealand presents just such an opportunity. Penguin nests have been counted across 48 sites and 15 years, and different restoration treatments were put in place at different sites at different times. The cost of each of these measures is known. Hence, we are able to compare the effectiveness and cost effectiveness of three commonly used endangered species restoration techniques—trapping of introduced predators, revegetation, and intensive management.

The yellow-eyed penguin

The yellow-eyed penguin, or hoiho, is the third largest of the penguin species, standing up to 65-70 cm and weighing 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 the southeast coast of the South Island, Stewart Island, Campbell Island, and Auckland Island. The yellow-eyed penguin depends on the ocean for food and on the land for nesting habitat. In prehistoric 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). Adult yellow-eyed penguins have 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). Juvenile yellow-eyed penguins are somewhat more mobile. Richdale (1957) found that 81% of birds returned to their place of hatching or to a nearby area to breed. Darby (1996) found that close to 90% of breeding birds fledged either in the immediate vicinity or within 500 meters of their natal area. 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 in September lay, most commonly, two eggs. Chicks hatch in early November and fledge in early February (Darby and Seddon 1990). Fledged chicks become juveniles, who head out to sea with no further parental supervision (Seddon, 1990). Juvenile mortality has an estimated mean of 52% (Richdale, 1957), but has been recorded as high as 88% at one site (Darby and Seddon, 1990).

Although penguins in general are highly appealing to humans (Davis and Renner, 2003), one study finds the yellow-eyed penguin to be less aesthetically pleasing than other penguins (Stokes, 2006). Nevertheless, viewing yellow-eyed penguins has been shown to elicit wonder, improve mood, and increase environmental awareness (Schanzel and McIntosh, 2000). Existence value for the yellow-eyed penguin is speculated to be high, though to our knowledge no contingent valuation study has been undertaken for this or any other penguin species.

A substantial nature tourism industry has developed around the yellow-eyed penguin. An estimated 126,000 tourists, or 5.7% of New Zealand's 2.2 million international visitors in 2006-2007, visited penguins while in New Zealand, though this figure also includes tourists who visited blue penguins and Fiordland crested penguins (Ministry of Tourism, 2007a). With the average holiday visitor spending NZ\$3115 in New Zealand (Ministry of Tourism, 2007b), penguins clearly add millions of dollars of value 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.

The yellow-eyed penguin faces a variety of threats. On land, chicks face predation from cats, dogs and mustelids (ferrets, stoats, and weasels), 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). 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’ (Birdlife International, 2005). The yellow-eyed penguin is one of the three most endangered penguin species along with the Galapagos Penguin (Birdlife 2007a) and the Erect-Crested Penguin (Birdlife International, 2007b), and 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 their 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. 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 Department of Conservation, the Yellow Eyed Penguin Trust, and private landowners and conservationists are all contributing effort towards the recovery of the species. These actors have implemented a diverse range of restoration treatments across sites and years. This paper evaluates the effects of three restoration treatments—trapping of introduced predators, revegetation, and intensive management—on 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 mammals to New Zealand, bird populations of many species were decimated or driven to extinction (O’Donnell, 1996). Today, non-native ferrets, stoats, feral cats, and dogs all contribute to the terrestrial mortality of yellow-eyed penguins (Hocken, 2005). To reduce terrestrial mortality of juveniles, a common yellow-eyed penguin restoration treatment is trapping of mustelids and feral cats. Diverse trapping methods have been used across years and sites. In some cases poison baited traps are set in lines across a property; in other cases traps are placed near known nests. In some cases traps are placed and checked at intervals throughout the year; in other cases traps are placed and maintained only during the nesting season

(MacFarlane, pers.comm.). This analysis does not distinguish between 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 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 restoration treatment is revegetation. Diverse revegetation methods have been used. In one method, all grassland is seeded with native trees and shrubs, and weeded several times in the following years until native bush can take hold. In another method, constructed nest boxes are placed in the pastoral landscape and surrounded by stands of flax (*Phormium tenax*) (MacFarlane, pers.comm.). This analysis does not distinguish between these two styles of revegetation.

Intensive management

Both adult and juvenile yellow-eyed penguins are impacted by disease, starvation, and trauma. To reduce these impacts, full time managers provide intensive management at several 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 from elsewhere. In addition to providing treatment for individual birds, managers can enhance predator control efforts, 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, pers.comm.).

Data

Our dependent variable is constructed from the New Zealand Department of Conservation yellow-eyed penguin nest counts. DOC has assembled a superb panel data set of yellow-eyed penguin nest counts during the 1992/1993-2006/2007 breeding seasons at 48 discrete sites along a 300 km stretch of the southeast coast of New Zealand's South Island (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 nests estimated after the three visits (McKinlay, pers.comm.). Although yellow-eyed penguin population data was also available from Banks Peninsula across six consecutive breeding seasons and six sites, this dataset was not used because this survey counted eggs, adults, and chicks rather than nests.

Each nest is occupied by 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 proportion is lower during exceptionally bad years (Efford *et al*, 1994). The total yellow-eyed penguin population is not used as a metric because juvenile mortality is high and variable. The total adult population is not used as this is more difficult to survey than nests.

We compiled a panel data set of restoration treatments used 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.comms). A site-year has a treatment variable of 1 if that treatment was applied during that site-year, and has a treatment variable of 0 otherwise. Some sites never received any treatment; other sites received a particular treatment in all years. Treatments went from 0 to 1 at many sites, but never went from 1 to 0. A Venn diagram of site-years receiving restoration treatments is shown in Figure 1.

Ecological theory predicts that site growth rate will 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 closest city, using Google Maps. Data on site land cover, slope, aspect, and distance from road are taken from the New Zealand Land Cover Database (Terralink, 2007). Percent of land cover in each vegetation type was determined for a 250m radius about the center of each site. Summary statistics are found in Table 1.

We estimated the marginal cost per hectare of implementing a restoration treatment following discussions with practitioners (B. McKinlay, A. Spencer, R. Goldsworthy pers.comm). These are displayed in Table 2. These figures represent what it would have cost DOC to provide a restoration treatment across one hectare at one site. Actual incurred expenditure by private groups on revegetation and intensive management was lower than the cost that would have been incurred by DOC, as these groups were able to use volunteer rather than paid labor. Overhead costs, for instance office costs and vehicle costs, are not included in the marginal cost per hectare, as it is assumed that these costs would have been borne whether or not a particular yellow-eyed penguin restoration treatment was undertaken at a particular site.

Method

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

$$\ln \lambda_{it} = \beta_0 + X_{it}' \beta_1 + \beta_2 \ln \delta_{it-1} + \beta_3 y_t + \varepsilon_{it} \quad (1)$$

Here the observation $\lambda_{it}=n_t/n_{t-1}$ represents the annual population growth rate in nests, n , at site i between year t and $t-1$. X is a matrix of dichotomous variables representing whether or not each restoration treatment was employed at site i in year t . δ_{t-1} represents

nest density (nests per hectare) at site i in year $t-1$. y_t is a year dummy. See Figure 2 for a timeline of restoration treatments and nest counts.

Population growth rate, λ , rather than nest numbers or absolute change in nest numbers, is the proper dependent variable for several reasons. First, logarithmic population growth rate λ_t is multiplicative rather than additive. λ_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 likely prone to underestimation and measurement error. It is likely that underestimation is more severe at sites which are more remote, more vegetated, steeper, or otherwise more difficult to survey (McKinlay, pers.comm.). Using population growth rate as the dependent variable neutralizes heterogeneity in measurement error across sites, since we assume that underestimation of nests occurs proportionally in both n_t and n_{t-1} .

Observations were omitted when either n_t or n_{t-1} were uncounted or were equal to zero, leaving 519 usable observations of λ_{it} . The natural log of λ_t is used as the dependent variable rather than λ_t because the mean of $\ln \lambda_t$ is symmetric around zero for periods of time in which growth rate is zero.

Ecological theory predicts that growth rate could diminish at a site as population approaches carrying capacity. Log of density is included as a regressor to account for this possibility, following the ecological model of density-dependent population growth, $\frac{dN}{dt} = rN(1 - \frac{N}{K})$ (Lotka, 1925). All regressions were performed using Stata. All regressions are OLS unless otherwise noted.

When density dependence is not included in the regression, no treatment is significantly correlated with increased population growth rate; see results in Table 3(a). Once the log of site density is included as a regressor, intensive management is significantly correlated with increased population growth rate; see Table 3(b). Explanatory power increases when year dummies are introduced to account for interannual fluctuations in penguin mortality and fecundity. These regression results are displayed in Table 3(c).

Potentially the effects of restoration treatments may not be visible immediately. An alternative specification allows for a temporal lag of three years, the length of time for newborn chicks to enter the adult breeding population.

$$\ln \lambda_{it} = \beta_0 + X_{it-3}' \beta_1 + \beta_2 \ln \delta_{it-1} + \beta_3 y_t + \varepsilon_{it} \quad (2)$$

Significance of treatments is robust to lags; results are displayed in Table 3(d).

As an alternative test of the effects of treatment over time, we regress log of growth rate on explanatory and control variables, over time increments of $k \geq 2$ years:

$$\ln \lambda_{it} = \beta_0 + \sum_{j=1}^k X_{it-j} \beta_1 + \beta_2 \delta_{it-k} + \beta_3 y_t + \varepsilon_{it} \quad (3)$$

We find that over any time scale, intensive management is significant, while trapping and revegetation are not. This is shown in Table 4.

Because adult yellow-eyed penguins exhibit strong nest site fidelity, and pre-breeding yellow-eyed penguins are largely philopatric as well, population change at the site level is a defensible dependent variable. However, some pre-breeding yellow-eyed penguins do move between sites, and the penguins' predators are mobile as well. A third specification allows for the possibility that a site could be influenced by restoration treatments implemented at nearby sites:

$$\ln \lambda_{it} = \beta_0 + X_{it} \beta_1 + W_{t-1} \beta_2 + \beta_3 \ln \delta_{it-1} + \beta_4 y_t + \varepsilon_{it} \quad (4)$$

Here matrix W represents the number of other sites within 10 km at which a particular restoration treatment was implemented. The effect of intensive management on growth rate falls just below significance in this specification; treatments within a 10 km neighbourhood were not significant. Results are displayed in Table 3(e).

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). 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. Results are displayed in Table 3(f). The sea lion is, as expected, significantly correlated with a reduction in the population growth rate. The magnitude of the intensive management coefficient increases after accounting for the sea lion. Other omitted variables are likely to impact penguin population. Penguin populations are known to have been affected in the past by fire and vandalism, for instance (McKinlay, pers.comm). However, because there is no complete record across sites and years of whether or not these activities occurred, these can not be included as control variables.

Selection bias and the instrumental variable approach

Site-years in which restoration treatments took place were not decided randomly. Sites receiving restoration treatment were selected opportunistically—a situation described by a representative of the Yellow Eyed Penguin Trust as an '*ad hoc*ocracy' (Kennedy, 2007). It is conceivable that site-year characteristics could be correlated with both penguin population growth rate and probability of treatment. If restoration treatments were more likely to be put in place at sites where growth rate is most steeply decreasing, then regression coefficients for the effect of treatments would be biased downward. If restoration treatments were more likely to be put in place at sites where growth rate is most steeply increasing, then regression coefficients for the effect of

treatments would be biased upward. We confront this potential endogeneity problem in three ways—control for observables, site fixed effects, and two stage least squares.

First, we include as regressors all observable site characteristics, including dummies for the three regions (North Otago, Otago Peninsula, Catlins), percent of site in each land cover type (sand and gravel, grassland, scrub and shrubland, and forest), mean slope, and mean elevation. Results are displayed in Table 3(g). The significance of intensive management is robust to the inclusion of control variables.

Next, we run a regression with site fixed effects. Results are displayed in Table 3(h). With a fixed effects model, the treatment effect is no longer identified from differences in growth rate across sites, but only from differences within sites across years. Figure 3 illustrates the decrease in data points when sites with intensive management in all years are removed from the identification strategy, leading to a decrease in the variance of the effect of intensive management.

Third, we construct an instrumental variable from variables related to site accessibility. These variables are correlated with probability of a conservation treatment occurring, but are plausibly uncorrelated with penguin growth rate. These variables are site distance from the major city of Dunedin, site distance from a road, mean slope of a site, and percent of site in grassland rather than shrubland, forest, or sand and gravel. None of these instrumental variables is significant when included in a regression of growth rate on explanatory variables (Table 3(g)). The correlation between these variables and probability of a conservation treatment occurring in a given site-year is shown in a SUR regression in Table 5.

Because there are three conservation treatments, we required at least three instruments. Logically the strongest instruments are site distance from Dunedin and site distance from a road. Sites closer to roads and closer to Dunedin are more accessible for conservation, but are not expected to be more or less conducive to penguin growth rates. Weaker instruments are mean slope of a site, and percent of a site in grassland. Flatter, grassier sites are expected to be more accessible for conservation. These site characteristics could be suspected to impact population growth rate, although no significant impact appears in the regression (Table 3(g)). Results from the two stage least squares regressions are displayed in Table 6. First stage F-statistics are below 10, so the instrument must be regarded as weak.

Cost effectiveness

In the second stage of analysis, the average cost-effectiveness of each restoration treatment is determined, using the formula:

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

Here CE_x is the average cost effectiveness of treatment x , that is, the average number of nests gained by 2006 per dollar spent on a treatment. N_{2006} is the actual number of nests across all sites in 2006. \hat{N}_{2006} is the counterfactual number of nests present in 2006, that is, the number of nest sites that would have been present in 2006 if a particular restoration treatment had not been undertaken anywhere from 1992-2006. C_x is 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 the actual expenditure on the treatment over this time period. Table 7 reports the average cost per site-year of providing each treatment.

Actual number of nests, N_{2006} , is equal to $\sum_i n_{i2006}$. Where n_{it} was not directly counted, it is predicted using the inverse of the specification in (2):

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

The result, $N_{2006} = 462.5$, differs only slightly from DOC's estimate of $N_{2006} = 464$, which was obtained using the prediction model $n_{it} = 0.95n_{it-1}$ (McKinlay, pers.comm.).

The counterfactual number of nests had no treatment been undertaken, \hat{N}_{2006} , is equal to $\sum_i \hat{n}_{i2006}$. When treatment x was not undertaken at a site, counterfactual nests equals observed nests, $\hat{n}_{it} = n_{it}$. When the treatment was undertaken at a site, we predict \hat{n}_{it} by subtracting the influence of the treatment from the actual growth rate observed during a particular site-year, using the model:

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

Here, $\hat{\beta}_x$ is the coefficient in the vector of coefficients $\hat{\beta}_1$ representing the effect of treatment x . To create a confidence interval around \hat{N}_{2006} at the 95% confidence level, we follow the model in (7), substituting $\hat{\beta}_{x.025}$ for $\hat{\beta}_x$ to create an upper bound, and substituting $\hat{\beta}_{x.975}$ for $\hat{\beta}_x$ to create a lower bound. Results are displayed in Table 8. We find that the marginal cost of producing an additional yellow-eyed penguin nest using intensive management is \$68,600 without accounting for the sea lion, or \$49,500 accounting for the sea lion. At the 95% confidence interval, neither trapping nor revegetation can produce an additional yellow-eyed penguin nest at a finite cost.

This prediction model has the advantage of predicting counterfactual growth rates using actual observed site-year specific errors. The drawback is that in this model counterfactual growth rates do not account for new counterfactual site densities, which we know to be significant predictors (Table 3(b-h)). However, including density in the predictive model, as in (6), would have a more significant drawback: counterfactual

population would be driven entirely by density. As the number of years of prediction increases at a site, site population would converge on a density dependent equilibrium level far more rapidly than observed naturally.

Results

Of the three restoration treatments analysed, only intensive management is significantly correlated with an increase in site-level yellow-eyed penguin population growth rate. This finding is robust to model specification and inclusion of control variables, but not to site fixed effect or two stage least squares. Intensive management was responsible for a 0.0523 average increase in log growth rate, or a 5.4% average increase in growth rate (Table 3(c)). This is equivalent to 0.67 average additional nests per site-year of treatment over all intensively managed site-years, or NZ\$68,600 per additional nest (Table 8).

When accounting for the presence of the sea lion at two intensively managed sites, the positive effect of intensive management on growth rate appears even stronger. When including 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 (Table 3(f)). 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 (Table 8).

At least two explanations have been advanced for the greater population growth observed in intensively managed site-years. One explanation is that treating sick or injured adults directly decreases adult mortality. Furthermore, treating sick, injured, or underweight penguins may allow adult penguins to devote more food energy to themselves rather than to their chicks, increasing the survivorship of breeding adults. This is especially true for the treatment of breeding females, and during low food years (Ratz, pers. comm.). That increased adult survivorship 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, and 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.

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, pers. comm.). 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. Future research into the movement patterns of translocated 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, at these sites, over the time period of this study, in the style in

which these treatments occurred. 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). We did not collect data on the particular style or the level of intensity with which treatments were implemented across sites. 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 manners of trapping or revegetation.

However, we are able to disaggregate the effect of treatments by the length of time a treatment has been in place at a site. There is reason to believe that length of time a treatment has been in place could influence the treatment's effect on growth rate. For example, higher nest densities have been recorded in the dense, low cover of scrub mosaics and early stage regenerating forest cover than in the relatively open understory of mature forest (Seddon, pers. comm.). This would imply that revegetation would have the greatest effect on growth rate during its early or middle years, and a smaller effect in later years. Table 9 shows the results of a regression which disaggregates treatments by years, using the following specification, where ψ represents the number of years a treatment has been in place:

$$\ln \lambda_{it} = \beta_o + X_{it|\psi \in [1,5]} \beta_1 + X_{it|\psi \in [6,10]} \beta_2 + X_{it|\psi \in [11,15]} \beta_3 + \beta_4 \delta_{it} + \beta_5 y_t + \varepsilon_{it} \quad (8)$$

Results in Table 9 show that the magnitude of the effect of revegetation on growth rate might be largest at late stages of succession (years 11-15), but the effect is insignificant at all stages. For trapping, it might be the case that penguin populations respond quickly to trapping in early years as the threat from predators is reduced, then level off again in middle or later years as a new, predator-free, carrying capacity is reached. This would imply that trapping would have the greatest effect on growth rate during the first five years, as larger numbers of chicks enter the breeding population. Results in Table 9 find that the magnitude of the effect of trapping on growth rate might be largest in middle years (years 6-10), but the effect is insignificant at all stages.

Though not effective in increasing yellow-eyed penguin population growth rate, revegetation and trapping 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), the mohua (O'Donnell *et al.*, 1996).

The presence of the New Zealand sea lion had a large and significant negative impact on penguin growth rate at the sites in which it was present (Table 3(f)). Lalas *et al* (2007) discuss potential management actions pertaining to this threat.

Notably, this analysis finds that yellow-eyed penguin population growth rate is negatively correlated with nest density (Table 3(b-h)). While this is consistent with 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.

The coefficient of determination, R^2 , increases from 0.05 to 0.19 following the inclusion of year dummies (Table 3(c-g)), 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). 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 factors may play a larger role in penguin mortality than predation; an examination of 124 opportunistically collected yellow-eyed penguin specimens 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 this study, while mustelid trapping has the potential to reduce a much smaller proportion of 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 *ex ante* planning. Conservationists are frequently faced with the challenge of analyzing the impact of treatments without the benefit of proactive study design. With enough data points, panel econometric techniques can be used to perform *ex post* analysis on the impact of treatments by effectively re-randomizing treatments.

When analysis is performed *ex post*, analysts must confront the possibility that treatments may not have been assigned across sites randomly. Locations for treatments may instead have been targeted to sites where they were expected to have the greatest impact, or based on ease of access. In this paper we have demonstrated three ways in which non-random assignment of treatments can be confronted. First, control variables for site characteristics can be included in the regression. Second, a site fixed effects model can account for variation at the individual site level. And finally, a two-stage least squares model with an instrumental variable constructed from data related to site accessibility can be used, when site accessibility is plausibly correlated with probability of treatment but not biological success. This instrument is only useful for species such as the yellow-eyed penguin, for which proximity to humans does not constitute a primary threat.

The impact of a treatment may not always occur during the year or at the site in which it takes place. When the length of time from treatment to impact is predictable, as in the case with reduced mortality of chicks and number of breeding adults, the dependent variable can be regressed on the lag of the explanatory variable. When the distance from treatment to impact is predictable, the dependent variable can be regressed on the number of occurrences of treatment within that distance from a site.

In this study we find that intensive management is positively correlated with increases in yellow-eyed penguin annual population growth rate, while trapping of predators and revegetation are not. This finding is robust to model specification and the inclusion of control variables, though not to site level fixed effects or to two stage least squares. This finding is also consistent with two yellow-eyed penguin population viability analyses, which show that growth rate is sensitive to changes in adult mortality, and with a necropsy study, which found that many penguins were dying of the stresses intensive management is designed to reduce.

Conservation groups with access to funding or volunteer labor might consider expanding intensive management to new sites. Not only is intensive management the most promising restoration treatment, but intensive management at new sites would provide additional data for identification for a fixed effects model identification. Expanded intensive management could be combined with research on which intensive management mechanisms are responsible for increased population growth rates. The estimated average cost of NZ\$68,600 per nest through intensive management could be considered low when compared to the estimated several millions in tourism revenue generated by viewing penguins annually found by Tisdell (2007).

This analysis did not differentiate across style or intensity of treatment at a site. We can only make predictions on expanding a treatment to a new site, not on changing the style or increasing the intensity of treatment at a site.

The methods employed here for evaluating effectiveness of yellow-eyed penguin restoration treatments can be extended to any conservation program anywhere, though a few key elements contribute to the success of the evaluation in this case. The cornerstone of a successful program evaluation is a comprehensive data set of a biological metric of success, in this case nest counts. The metric should be a robust indicator of population size or health, and ideally should be easy and cheap to ascertain. The metric should be monitored regularly using consistent methodology across all sites. Observations of biological success must be made at the level of independent populations. We were fortunate in this case that yellow-eyed penguin populations live at many sites, and are stationary from year to year, with little exchange of members between sites.

Unless the species is in critical danger, managers should leave some sites as controls, deliberately withholding restoration treatments. These control sites are critical to determining the counterfactual—what would have happened at a site if no management were undertaken. This evaluation benefited from the multitude of control sites, as well as

from the diversity of management measures employed by different conservation actors across sites. Analysis is easiest when treatment and control sites are randomly selected, though as we have seen, panel econometrics can compensate for non-random site selection.

This analysis relied upon indicative estimates of costs rather than actual expenditure records. We were unable to use data from DOC on historical expenditures since these data were not disaggregated by restoration treatment, nor did they separate overhead costs from marginal costs. Keeping disaggregated accounts of expenditures is a key component of accurately determining conservation cost effectiveness.

A program of monitoring, control sites, and cost tabulation allows analysis of which conservation treatments provide best value for money. Even an ineffective program becomes valuable if we can learn from its mistakes. On the other hand, a treatment which has been shown to be cost effective should attract further resources for investment. We submit the case of the yellow-eyed penguin with the hope that effectiveness and cost effectiveness analysis will be extended to conservation in other settings.

Acknowledgments

Compiling a comprehensive data set on which treatments were put in place when at which sites would not have been possible without discussions with David Agnew, Rosalie Goldsworthy, David MacFarlane, Euan Kennedy, Chris Lalas, Hiltrun Ratz, Phil Seddon, and Fergus Sutherland. Helpful comments on early versions of the manuscript were provided by Chris Costello, Frank Davis, Bob Deacon, Olivier Deschenes, Paul Ferraro, Hiltrun Ratz, Phil Seddon, Doug Steigerwald, and seminar participants at Lincoln University and the University of California, Santa Barbara. Brad Case provided GIS analysis. Busch gratefully acknowledges financial support from NSF IGERT grant number 0114437, and from the NSF East Asia Pacific Summer Institute.

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 (2005). *Megadyptes antipodes*. In: IUCN 2006. *2006 IUCN Red List of Threatened Species*. <www.iucnredlist.org>.
- Birdlife International (2007a). *Spheniscus mendiculus*. In: IUCN 2007. *2007 IUCN Red List of Threatened Species*. <www.iucnredlist.org>.
- Birdlife International (2007b). *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 to 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.
- Davis, L.S. and M. Renner (2003). *Penguins*, Yale University Press, New Haven, Connecticut, 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, pubs., 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, *et al.* 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.

Schanzel and 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.

Stokes, D.L. (2006). Things we like: Human preferences among similar organisms and implications for conservation, *Human Ecology*, 35:361-369.

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 *et al* (2007). Conserving biodiversity efficiently: What to do, where, and when. *PLoS Biology*, 5(9).

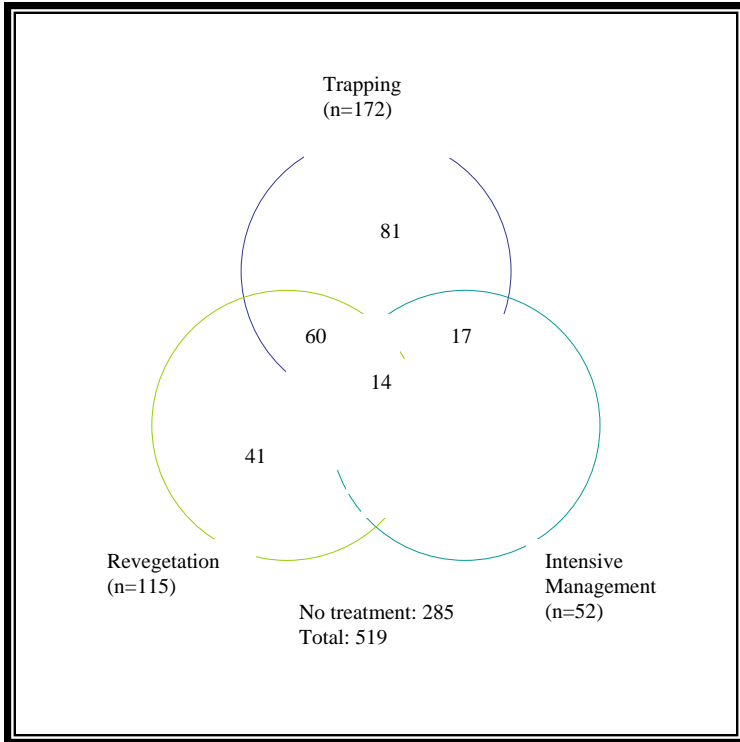


Figure 1 – Number of site-years receiving restoration treatments

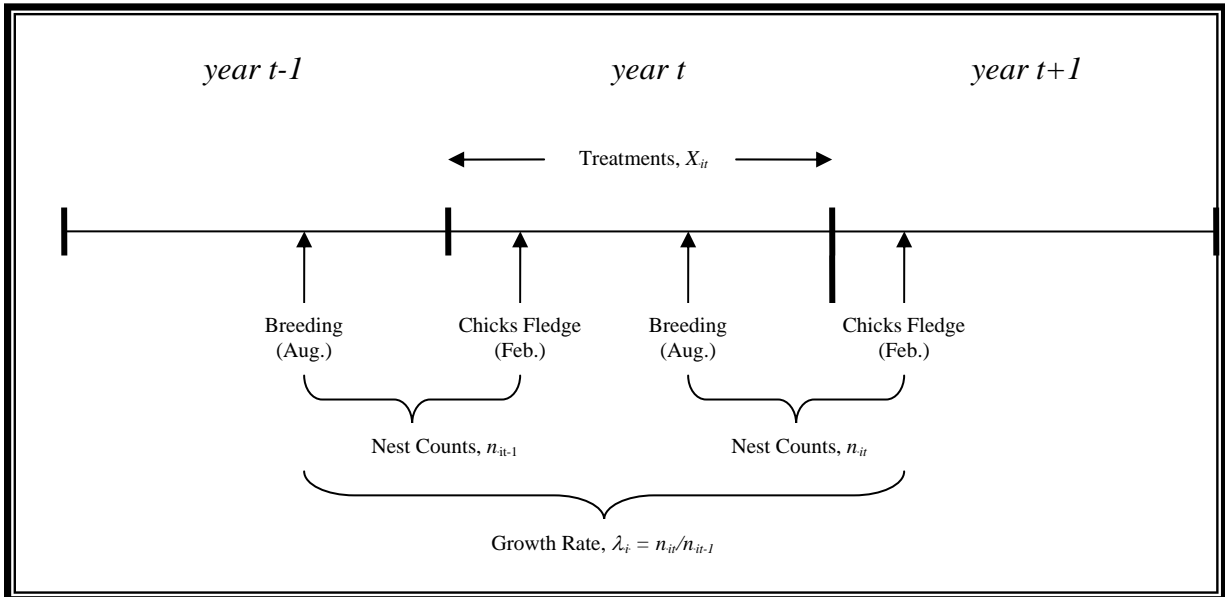


Figure 2 – Timeline of restoration treatments and nest counts

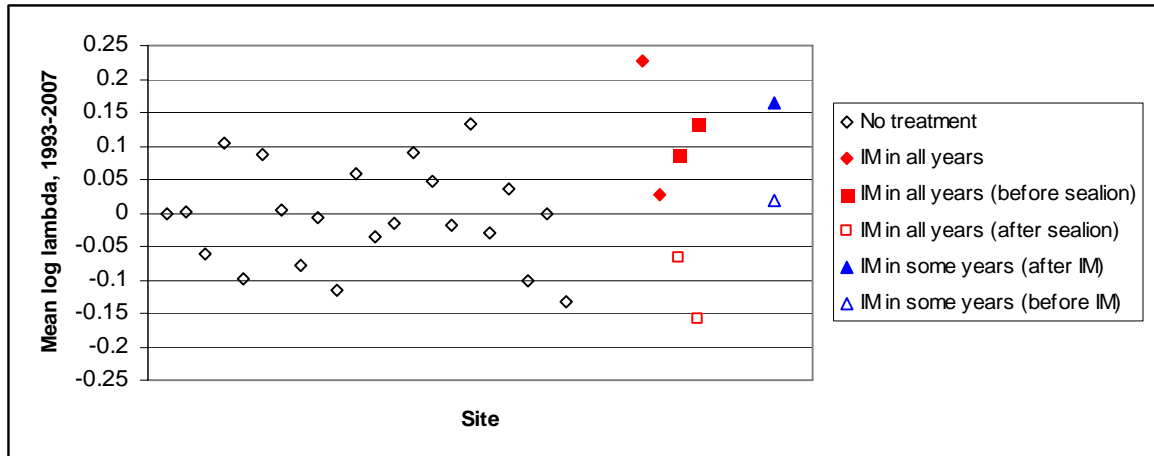


Figure 3 – Effect of intensive management on average growth rate at sites

	n	min	max	median	mean	st. dev
Year	15	1992	2006	-	-	-
Site	48	-	-	-	-	-
Population(site i, year t)	606	0	55	8	10.80	10.33
Population>0(site i, year t)	540	1	55	10	12.12	10.19
lambda	519	0.143	7	1	1.097	0.541
log lambda	519	-0.845	0.845	0	0.002	0.179
trapping	672	0	1	0	0.275	0.447
revegetation	672	0	1	0	0.184	3.882
intensive management	672	0	1	0	0.077	0.267
sealion	672	0	1	0	0.030	0.170
area (Ha)	644	1	25	5	6.989	5.541
Density (nests/Ha)	585	0	21	1.5	2.159	2.234
Grassland (%)	672	0	1	0.739	0.646	0.315
Shrubland (%)	672	0	0.577	0	0.049	0.126
Forest (%)	672	0	0.971	0	0.113	0.230
Sand and Gravel (%)	672	0	0.803	0.104	0.171	0.199
Other (%)	672	0	0.423	0	0.020	0.072
Mean Elevation (m)	672	1.879	289.020	30.712	39.611	44.518
Mean Aspect (deg)	672	55.705	310.409	137.385	138.994	49.393
Mean Slope (%)	672	0.763	37.907	11.662	12.521	8.229
Distance from Dunedin (km)	672	12.6	196	71.9	81.663	59.446
Distance from Road (m)	672	14.106	3864.805	787.917	961.951	855.803

Table 1 – Summary Statistics

Revegetation	First Year		Years 2, 4, 6	
Materials	\$3 2500 \$7,500	per plant plants per hectare per hectare		
Labor	\$16.00 4.5 50 \$3,600.00	per person-hour hours people per hectare	\$16.00 8 4.5 \$576.00	per person hour hours people per hectare
Total	\$11,100.00	per hectare	\$576.00	per hectare
Traplines	First Year		Each Subsequent Year	
Materials	\$50 5 \$12.50 5 \$312.50	per trap traps per hectare bait and poison per trap per year traps per hectare per hectare	\$50 0.5 \$12.50 5 \$87.50	per trap traps per hectare bait and poison per trap per year traps per hectare per hectare
Labor	\$16.00 1 52 \$832.00	per person-hour hour per hectare per week weeks per year per hectare	\$16.00 1 52 \$832.00	per person-hour hours per week weeks per year per hectare
Total w/o intensive management (Materials plus Labor)	\$1,144.50	per hectare	\$919.50	per hectare
Total w/ 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/site+\$50/nest		\$40,000/site+\$50/nest	

Table 2 – Average cost of restoration treatments per hectare

	(a)	(b)	(c)	(d)	(e)	(f)	(g)	(h)
n	519	506	506	506	506	506	506	506
Intercept	0.0007 (0.07)	0.02013* (1.89)	0.0478 (1.51)	0.0516* (1.65)	0.0508 (1.36)	0.0475 (1.50)	0.0937 (1.59)	0.1476*** (3.91)
Trapping	0.0091 (0.50)	0.0096 (0.54)	0.0151 (0.89)	-0.0055 (-0.31)	0.0253 (1.29)	0.0116 (0.68)	0.0133 (0.67)	0.0013 (0.04)
Revegetation	-0.0140 (-0.69)	-0.0041 (-0.21)	-0.0107 (-0.57)	0.0164 (0.82)	-0.0194 (-0.95)	-0.0050 (-0.26)	-0.0058 (-0.28)	-0.0047 (-0.12)
Intensive Management	0.0185 (0.69)	0.0582** (2.13)	0.0523** (2.03)	0.0478* (1.72)	0.0419 (1.52)	0.0847*** (2.64)	0.0815** (2.40)	0.0807 (0.79)
Log Density	no	-0.1104*** (-5.12)	-0.0998*** (-4.82)	-0.1002*** (-4.84)	-0.1039*** (-4.91)	-0.1009*** (-4.89)	-0.1185*** (-5.11)	-0.5376*** (-12.93)
Year Effects	no	no	yes	yes	yes	yes	yes	yes
Three year lag	no	no	no	yes	no	no	no	no
Treatments within 10 km	no	no	no	no	yes	no	no	no
Sea lion	-	-	-	-	-	-0.0813* (-1.69)	-0.858* (-1.66)	-0.0944 (-1.46)
Otago Peninsula	-	-	-	-	-	-	-0.0098 (-0.29)	-
Catlins	-	-	-	-	-	-	-0.0049 (-0.14)	-
Grassland (%)	-	-	-	-	-	-	-0.0181 (-0.41)	-
Shrubland (%)	-	-	-	-	-	-	-0.0151 (0.19)	-
Forest (%)	-	-	-	-	-	-	-0.0387 (-0.74)	-
Mean Elevation (m)	-	-	-	-	-	-	-0.0002 (-0.94)	-
Mean Slope (%)	-	-	-	-	-	-	0.0003 (0.19)	-
Distance from Dunedin (km)	-	-	-	-	-	-	-1.74x10 ⁻⁴ (-0.43)	-
Distance from road (m)	-	-	-	-	-	-	-1.30x10 ⁻⁶ (-0.08)	-
Site Fixed Effects	no	no	no	no	no	no	no	yes
R²	0.0023	0.0519	0.1898	0.1876	0.1921	0.1945	0.2009	0.0918 [#]
Adjusted R²	-0.0036	0.0443	0.1616	0.1593	0.1588	0.1647	0.1558	-

Table 3 – Effect of restoration treatments on site-year 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. Within R²=0.3859; Between R²=0.0370.

Increment	1	2	3	4	5	6	7
n	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 Effects	yes	yes	yes	yes	yes	yes	yes
R ²	0.1898	0.2811	0.3377	0.3148	0.2822	0.3066	0.3340

Table 4 – Effect of restoration treatments on site-increment growth rate, subsampled in multiple year increments

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
N	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 Effects	yes	yes	yes	yes	yes	yes	n.a.
R ²	0.3197	0.2947	0.2236	0.2189	0.2613	0.2741	0.2758

Table 4 (cont'd.) – Effect of restoration treatments on site-increment growth rate, subsampled in multiple year increments

OLS regression; t-statistic in parentheses

*Significant at $p < 0.10$

**Significant at $p < 0.05$

***Significant at $p < 0.01$

	Trapping	Revegetation	Intensive Management
n	672	672	672
Intercept	-0.0052 (0.09)	0.2033*** (3.92)	0.1276*** (3.68)
Distance from Dunedin (km)	-0.0145*** (-5.11)	-0.0001 (-0.46)	-0.0009*** (-5.15)
Distance from Road (m)	2.57×10^{-5} (1.29)	1.57×10^{-5} (-0.85)	1.32×10^{-5} (1.07)
Mean Slope (%)	0.151*** (5.71)	-0.0031 (-1.29)	-0.0016 (-0.99)
Grassland (%)	0.3967*** (7.43)	0.1019** (2.08)	0.1016*** (3.10)
Mean Elevation (m)	-0.0020*** (-4.08)	-0.0005 (-1.13)	-0.0009*** (-2.81)
Pseudo R²	0.1266	0.0210	0.0770

Table 5 – Likelihood that a restoration treatment will be taken

Seemingly unrelated regression (linear); z-score in parentheses

*Significant at $p < 0.10$

**Significant at $p < 0.05$

***Significant at $p < 0.01$

	(a)	(b)
n	506	506
Intercept	0.0333 (1.12)	0.0432 (1.12)
Trapping	-0.0145 (-0.27)	-0.0259 (-0.40)
Revegetation	-0.0878 (-0.57)	-0.1156 (-0.60)
Intensive Management	0.2781 (1.30)	0.3883 (1.07)
Log density	-0.1481*** (-3.57)	-0.1389*** (-3.19)
Sea Lion	No	-0.3313 (-1.18)
Used as Instruments:		
-Distance from Dunedin	Yes	Yes
-Distance from road	Yes	Yes
-Mean Slope	Yes	Yes
-Grassland (%)	Yes	Yes
Root MSE	0.1907	0.1965
F statistic	5.26	3.90

Table 6 – Effect of restoration treatments on site-year growth rate: Instrumental Variable Approach

Two stage least squares; t-statistic in parentheses

*Significant at $p < 0.10$

**Significant at $p < 0.05$

***Significant at $p < 0.01$

	Trapping	Revegetation	Intensive Management	Total
Total Cost	\$1,456,667	\$1,344,252	\$2,619,350	\$5,420,269
Number of site-years	204	140	57	272
Average cost per site-year	\$7,141	\$9,602	\$45,954	\$19,927

Table 7 – Average cost of restoration treatment per site-year (NZ\$)

	Prediction Model	3(c)—Without Sea Lion			3(f)—Including Sea Lion		
	Actual nests, N_{2006}	462.5			462.4		
	Coefficient of magnitude of treatment	$\hat{\beta}_{.025}$	$\hat{\beta}_{.5}$	$\hat{\beta}_{.975}$	$\hat{\beta}_{.025}$	$\hat{\beta}_{.5}$	$\hat{\beta}_{.975}$
Trapping	Counterfactual nests, 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
	Nests gained from treatment (per site-year)	-0.30	0.20	0.54	-0.37	0.16	0.51
	Nests gained from treatment (per NZ\$100,000)	-4.19	2.84	7.57	-5.16	2.23	7.18
	Marginal cost of an additional nest (NZ\$)	Inf.	\$35,188	\$13,207	Inf.	\$44,823	\$13,927
Revegetation	Counterfactual nests, 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
	Nests gained from treatment (per site-year)	-0.69	-0.13	0.26	-0.60	-0.06	0.31
	Nests gained from treatment (per NZ\$100,000)	-7.23	-1.32	2.73	-6.23	-0.60	3.26
	Marginal cost of an additional nest (NZ\$)	Inf.	Inf.	\$36,628	Inf.	Inf.	\$30,691
Intensive Management	Counterfactual nests, 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
	Nests gained from treatment (per site-year)	0.03	0.67	1.04	0.33	0.93	1.24
	Nests gained from treatment (per NZ\$100,000)	0.06	1.46	2.26	0.71	2.02	2.69
	Marginal cost of an additional nest (NZ\$)	\$1,746,233	\$68,569	\$44,321	\$140,827	\$49,516	\$37,207

Table 8 – Average cost effectiveness of restoration treatments

	(a)	(b)	(c)	(d)	(e)
n	506	506	506	506	506
Intercept	0.0478 (1.51)	0.0489 (1.55)	0.0453 (1.44)	0.0517 (1.63)	0.0505 (1.60)
Trapping	0.0151 (0.89)	-	0.0105 (0.63)	0.0153 (0.90)	-
Revegetation	-0.0107 (-0.57)	-0.0107 (-0.57)	-	-0.0106 (-0.56)	-
Intensive Management	0.0523** (2.03)	0.0555** (2.16)	0.0522** (2.00)	-	-
Log Density	-0.0998*** (-4.82)	-0.0998*** (-4.80)	-0.1023*** (-4.91)	-0.0960*** (-4.71)	-0.0993*** (-4.79)
Year Effects	yes	yes	yes	yes	yes
Early Trapping (Year 1-5 only)	-	0.0168 (0.71)	-	-	0.0078 (0.33)
Middle Trapping (Year 6-10 only)	-	0.0237 (0.99)	-	-	0.0164 (0.68)
Late Trapping (Year 11-15 only)	-	0.0004 (0.01)	-	-	-0.0032 (-0.10)
Early Revegetation (Year 1-5 only)	-	-	-0.0021 (-0.07)	-	-0.0041 (-0.14)
Middle Revegetation (Year 6-10 only)	-	-	0.0030 (0.10)	-	-0.0000 (-0.00)
Late Revegetation (Year 11-15 only)	-	-	0.0120 (0.39)	-	0.0231 (0.75)
Early Intensive Management (Year 1-5 only)	-	-	-	0.0511 (0.86)	0.0555 (0.92)
Middle Intensive Management (Year 6-10 only)	-	-	-	0.0628 (1.30)	0.0661 (1.33)
Late Intensive Management (Year 11-15 only)	-	-	-	0.0603 (1.32)	0.0651 (1.32)
R²	0.1898	0.1907	0.1896	0.1896	0.1904

Table 9 – Effect of restoration treatments on growth rate, by time stage of treatment (OLS)