

Population viability analysis of New Zealand sea lions, Auckland Islands, New Zealand's sub-Antarctics: assessing relative impacts and uncertainty

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Abstract A common issue faced in wildlife management is how to assess the uncertainty of potential impacts on the viability of a species or population. The pup production of New Zealand (NZ) sea lions (*Phocarctos hookeri*) has declined 50% in the last 12 years at their main breeding area, the Auckland Islands. The two major known atypical impacts on NZ sea lions are as follows: (1) the direct mortality as bycatch of trawling and (2) bacterial epizootics, which can affect reproduction and mortality. Both of these impacts include high levels of uncertainty, with fisheries data being variable due to percentage observer coverage and the effect of sea lion exclusion devices, while the timing and severity of bacterial epizootics are not predictable. In this paper, an age-structured model of the NZ sea lion population at the Auckland Islands was built to examine the predicted effects of fisheries mortality and catastrophes (bacterial epizootics), both separately and then combined, on population viability over a 100-year period using the VORTEX population viability analysis programme. These models are then compared against 15 years of empirical field data to determine the actual level of impacts being observed. Model results indicate that although naturally occurring epizootics reduce the growth rate of the population, it does not cause a decline in the Auckland Island population. However, sustained fisheries bycatch at current estimated levels, particularly considering its potential impact on adult female survival, could result in a population decline and possible functional extinction over the modelled time period.

Keywords *Phocarctos hookeri* · Management · Fisheries interactions · Population viability analysis · Bacterial epizootic

Introduction

Understanding the causal factors responsible for the decline of long-lived, slow-reproducing species can be difficult because a long time-series of data are needed to assess and identify impacts and trends. This is particularly true for marine mammal species which can be difficult to study due to their aquatic habitat and often have small and isolated populations, i.e. Maui's Dolphins (*Cephalorhynchus hectori maui*, Baker et al. 2002), Galapagos fur seals (*Arctocephalus galapagoensis*, Trillmich and Limberger 1985) or Hawaiian monk seals (*Monachus schauinslandi*, Gerrodette and Gilmartin 1990). Small and restricted distributions of populations in any species make them highly susceptible to localised catastrophic events, hunting or bycatch pressure and disease epizootics. In pinnipeds, (seals, sea lions and walruses) bycatch pressure and disease epizootics are two recognised negative impacts on many species (Kennedy et al. 2000; Härkönen et al. 2006; Read et al. 2006). The global bycatch of pinnipeds is in the hundreds of thousands with bycatch likely to be having significant demographic effects on many populations (Read et al. 2006). Catastrophic disease events have been known to kill tens of thousands of pinnipeds in short periods of time and researchers still have limited knowledge for cause, severity or spread (i.e. Kennedy et al. 2000; Härkönen et al. 2006; Wilkinson et al. 2006).

Population viability analysis (PVA) is a quantitative approach to assessing casual factors affecting species population viability, particularly when long-term data series of a

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species is not available (Beissinger and Westphal 1998). PVA can be applied to estimate probabilities of populations' survival under different conditions using life history data and knowledge about the influence of environmental and external factors (Gamlimberti et al. 2001). However, it is rare to be able to test these models' predictions with empirical field data.

New Zealand sea lions (NZ sea lion, *Phocarctos hookeri*) are a species with a small, highly localised population that has declining pup production (Wilkinson et al. 2006; Chilvers 2008). The NZ sea lion is classified as 'Vulnerable' and projected in decline by the International Union for the Conservation of Nature (IUCN 2008) and 'Nationally Critical' under the New Zealand threat classification system (the most at risk classification within the system, Baker et al. 2010). Pup production at their main breeding area (Auckland Islands) has declined by 50% in the last 12 years (Chilvers et al. 2007; Chilvers 2011). The two major known atypical impacts on the NZ sea lion species are direct mortality as bycatch of trawling and bacterial epizootics, which can affect both reproduction and mortality. Both impacts include high levels of uncertainty in severity and occurrence. Therefore, PVA modelling is used here as a method to examine what factors and at what impact levels are critical and likely to be causing the decline of the NZ sea lion species. The species main breeding area, the Auckland Islands, also has 15 years of empirical field data (1995–2010), which is available to test the PVA models' predictions against, a scenario not usually possible for a long-live species.

The most significant known anthropogenic impact on NZ sea lions is the deaths of juveniles and adults as bycatch in the arrow squid (*Nototodarus sloanii*) and scampi (*Metanephrops challengeri*) trawl fisheries and the possible indirect impacts of resource competition and habitat modification caused by the trawl fishery (Wilkinson et al. 2003; Chilvers 2008; Robertson and Chilvers 2011). Since 2001, sea lion exclusion devices (SLEDs) have been used within the sub-Antarctic squid trawl fishery that operates around the Auckland Island shelf, February to May/August each year (Fishing area referred to as SQU6T: Chilvers 2008). A SLED is a metal separation grid fixed inside the trawl net at a 45° angle to the water flow, just before the cod end of the net (the collection area at the end of the trawl net that holds the captured target species, i.e. squid). SLEDs allow smaller objects, such as squid, to pass through the metal grid into the cod end, while larger objects, such as sea lions, should not pass but may escape via a hatch opening at the top of the net. Between 2004 and 2010, all fishing vessels used a SLED during fishing in SQU6T. There is significant uncertainty about the efficacy of SLEDs, and their use prevents direct counts of the number of sea lions harmed or killed in the fishery (Wilkinson et al. 2003; Chilvers 2008).

The other atypical factor for consideration in monitoring and predicting NZ sea lions population abundance and trends is the influence of naturally occurring epizootic events. There were no epizootics recorded until 1998/1999. Since then, NZ sea lions have been affected by two epizootics, one in 1997/1998 thought to cause by a *Campylobacter* bacteria and one lasting 2 years 2001/2002 and 2002/2003, known to be caused by *Klebsiella pneumoniae* (Wilkinson et al. 2006). These epizootics produce variable additional pup mortality in their first months of life, and the 1997/1998 epizootic resulted in adult deaths (Wilkinson et al. 2003, 2006). The epizootic events have not been linked with any natural or anthropogenic influences. The bacteria implicated in these events are not known to have come from human or domestic animals (P. Duignan pers. comm.), and there is no apparent link with timing of events and environmental factors such as El Niño/La Nina cycles or the Southern Oscillation (L. Meyneir, pers. comm.). This inability to determine when and what bacterial epizootics occur and what part or proportion of the population will be affected makes predicting the population trend of the NZ sea lion species difficult.

In this paper, an age-structured PVA of the Auckland Island NZ sea lion population is run, which considers a range of scenarios of varying impacts of bycatch and epizootics to make quantitative predictions of population size and likelihood of extinction over a 100-year period. These models are then tested by comparing the predictions from the first 15 years of these models (1995–2010) with empirical field data from the NZ sea lion Auckland Islands population from the same time period to determine the likely levels of impacts that have been occurring.

Materials and methods

Species and study area

NZ sea lions are long-lived, slow-breeding, polygamous colonial breeders, with recognised breeding areas found only in the New Zealand sub-Antarctics (Chilvers et al. 2007; Chilvers and Wilkinson 2009; Chilvers et al. 2010; Fig. 1). NZ sea lions have three breeding areas: (1) Northern Auckland Islands, Enderby and Dundas Islands (69% of pup production 2009/2010, Fig. 1), (2) Figure of Eight Island, Auckland Islands (2% pup production 2009/2010, Fig. 1) and (3) Campbell Island (29% pup production 2009/2010, Fig. 1; Chilvers et al. 2007; Chilvers 2011; Maloney et al. in press). For NZ sea lions, lactation lasts approximately 10 months from mid-December to October (Gales 1995), during which females split their time between foraging at sea within 175 km of breeding sites and spending time ashore feeding their dependent pup (Chilvers et al. 2005;

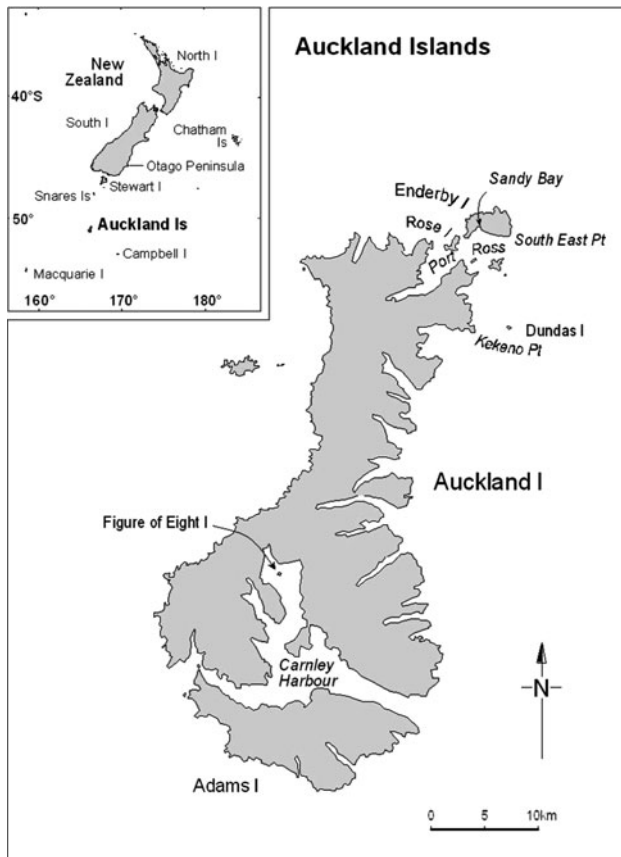


Fig. 1 Auckland Islands showing the main breeding areas for New Zealand sea lions: Sandy Bay and South East Point, Enderby Island, Dundas Island, 8 km south, and Figure of Eight Island, Carnley Harbour. *Inset:* New Zealand's sub-Antarctic showing the Auckland Islands and Campbell Island/Motu Ihupuku, where together over 99.9% of all NZ sea lion breeding occurs

Chilvers 2009). Most males (both breeding and non-breeding) disperse to the extremes of the species' range (up to 750 km north and south of the Auckland Islands) at the end of female oestrous in late January (Robertson et al. 2006).

PVA models

The PVA was run using VORTEX (version 9.93), a Monte Carlo simulation of the effects of deterministic forces as well as demographic, environmental events on wildlife populations (Miller and Lacy 2005). Simulation parameters are summarised in Table 1. Each simulation started in 1995 with a population estimate of 2,518 pups born at the Auckland Islands that year and population estimate of 11,200 (Gales and Fletcher 1999). This year was chosen as a starting point as it was the first year when reliable estimates of pup production were collected for all breeding areas of the Auckland Islands (Table 2 in Breen and Kim 2006) and the first absolute population estimate for the species (Gales and Fletcher 1999).

Juvenile and adult survival estimates (excluding the epizootic years) and environmental variability (EV, survival estimate 95% confidence intervals) were taken from Chilvers and Mackenzie (2010), female reproductive rate and EV were taken from Chilvers et al. (2010) and male reproductive parameters were from Robertson et al. (2006). Environmental variation is the fluctuation in population parameters resulting from annual fluctuations in the environment. Weather, the abundances of prey and predators, or the availability of breeding sites or mates can all vary, randomly or cyclically, over time; therefore, these naturally occurring variances are taken into account using EV parameters.

The sensitivity of the PVA to differing frequency and severity of bacterial mortality events, fisheries impacts and their combined effect were examined. Bacterial epizootics were modelled as levels of catastrophes. All catastrophes resulted in only 75% of normal (non-epizootic) survival for pups, this is the average decrease in pup survival from the 1998 and 2002/2003 epizootic events (Wilkinson et al. 2006). Therefore, the first two levels of catastrophe had pup survival decrease to 75% of normal (non-epizootic) on average, once every 5 or 10 years (Table 1). The highest level of catastrophe occurred at a average of once every 10-year interval and had pup survival decrease to 75% of normal (non-epizootic) and a 5% increase in mortality of all adults which is double that seen in the 1998 epizootic; however, all indications are that the 2.5% deaths were a minimum estimate (Table 1, Baker 1999).

Fisheries bycatch mortality was modelled as two levels of bycatch and at two sex ratios of animals' bycaught, giving four harvest levels (Table 1). Harvest levels 1 and 2 had a bycatch level of 89 NZ sea lions captured per year in SQU6T with sex ratios of (1) 58% female/42% male (average bycatch sex ratio for NZ sea lions from arrow squid fishery from 1995 to 2007, Table 2 in Chilvers 2008) and (2) 82% female/18% male (average bycatch sex ratio for NZ sea lions since SLEDs were introduced to the arrow squid fishery and used consistently by 100% of the fleet 2004–2009, Table 2, Robertson and Chilvers 2011). As observed on fishing vessels, bycatch mortality was imposed, at 75% of bycatch mortality on adults/25% on juveniles (Aoife Martin, MFISH pers. com.). The estimate of 89 is the average estimated number of deaths annually of NZ sea lions in SQU6T (both arrow squid and scampi fisheries) calculated by the NZ Ministry of Fisheries between 1995–1996 and 2009–2010 taking into account the use of SLEDs since 2004 (Aoife Martin, MFISH pers. com.), Baird (1996, 1999, 2005a, b) Baird and Doonan (2005), Smith and Baird (2005), the Ministry of Fisheries 2006–2007 SQU 6T Sea lion Operational Plan Initial Position Paper 2006 (<http://www.fish.govt.nz/NR/rdonlyres/8AE21896-401E-4AEC-BA69-AAE3B9923E1E/1241/200607FinalSQU6TIPP14sept1.pdf>, Thompson et al. 2010). Levels

Table 1 Demographic and other input values used for the VORTEX PVA program

Parameter	Value used (EV)	Source
<i>A. Demographic parameters</i>		
Female breeding age (years)	6	Chilvers et al. (2010)
Females maximum breeding age (years)	20	Confirmed breeding from known aged female Chilvers et al. (2010)
Male breeding age (years)	9	Cawthorn et al. (1985), Robertson et al. (2006)
Males maximum breeding age (years)	17	McNally and Chilvers pers comm.
Female mortality (%)		Chilvers and Mackenzie (2010)
Age 0 (excluding mass mortality years)	45% (8)	
Age 1	30% (8)	
Age 2	18% (6)	
Age 3	10% (4)	
Age 4	4% (2)	
Age 5	4% (2)	
Adults	2% (1)	
Male mortality (%)		Chilvers and Mackenzie (2010)
Age 0 (excluding mass mortality years)	64% (14)	
Age 1	35% (8)	
Age 2	14% (6)	
Age 3	8% (2)	
Age 4	2% (1)	
Age 5	2% (1)	
Age 6	2% (1)	
Age 7	2% (1)	
Age 8	2% (1)	
Adults	2% (1)	
Fecundity (% adult females pupping each year), EV in parenthesis	65 (10)	Chilvers et al. (2010)
Max litter size	1	Marlow (1975)
Birth sex ratio proportion of male	0.51	
Mating system	Polygamous	Chilvers et al. (2005), Robertson et al. (2006)
Males in breeding pool (%)	23	Robertson et al. (2006) (Based on 202 adult males bleached in season and 47 males maximum holding territory 47/202 = 23%)
<i>B. Simulation parameters</i>		
Initial population size (1995)	2,518 pups, total population estimate 11,200	Gales and Fletcher (1999), Chilvers et al. (2007)
Number of replications	500	Miller and Lacy (2005)
Number of years	100	Miller and Lacy (2005)
Catastrophes	1. ↑ pup mortality 25%/5 years 2. ↑ pup mortality 25%/10 years 3. ↑ pup mortality 25% and ↑ all adult mortality 5%/10 years	Baker (1999), Wilkinson et al. (2006), Chilvers et al. (2007)
Harvest/bycatch	1. 89/year sex ratio 58♀:42♂ 2. 89/year sex ratio 82♀:18♂ 3. 105/year sex ratio 58♀:42♂ 4. 105/year sex ratio 82♀:18♂	Ministry of Fisheries (see text for references) Thompson et al. (2010)
Quasi-extinction definition	<1,000 individuals	Pimm (1991), Traill et al. (2007)

EV Environmental Variation

Table 2 Predictions of models 1–13 for Auckland Island NZ sea lions, including mean annual exponential rate of population change across the simulations ($r \pm SD$), probability of quasi-extinction over100 years, the mean final extant population size ($\pm SE$) and the mean time (years) to quasi-extinction

Model	Description	Mean population change (r)	P-quasi-extinction	Mean extant population	Mean quasi-extinction time
1	Baseline	0.007 \pm 0.051	0	25,593 \pm 9,407	–
2	Baseline + pup mortality \uparrow 25% once every 10 years	0.005 \pm 0.051	0	19,378 \pm 8,026	–
3	Baseline + pup mortality \uparrow 25% once every 5 years	0.003 \pm 0.051	0	15,736 \pm 6,714	–
4	Baseline + pup mortality \uparrow 25% and adult mortality \uparrow 5% once every 10 years	0.000 \pm 0.055	0	12,895 \pm 5,981	–
5	Baseline + fishing mortality 89/year at 58♀:42♂ ratio	–0.003 \pm 0.053	0.028	10,534 \pm 6,929	90
6	Baseline + fishing mortality 89r 82♀:18♂ ratio	–0.019 \pm 0.055	0.562	5,836 \pm 4,422	80
7	Baseline + fishing mortality 105 58♀:42♂ ratio	–0.014 \pm 0.056	0.346	7,107 \pm 5,551	85
8	Baseline + fishing mortality 105 82♀:18♂ ratio	–0.029 \pm 0.059	0.872	4,658 \pm 3,053	73
9	Baseline + fishing mortality 89/year 58♀:42♂ ratio and pup mortality \uparrow 25% once every 5 years	–0.012 \pm 0.055	0.192	6,060 \pm 4,401	88
10	Baseline + fishing mortality 8/year 58♀:42♂ ratio and \uparrow pup mortality 25% and \uparrow adult mort 5%/10 years	–0.019 \pm 0.060	0.478	4,218 \pm 3,071	83
11	Baseline + fishing mortality 89 years 82♀:18♂ ratio and pup mortality \uparrow 25% once every 5 years/5 years	–0.028 \pm 0.058	0.856	4,085 \pm 3,425	75
12	Baseline + fishing mortality 89 years 82♀:18♂ ratio and \uparrow pup mortality 25% and \uparrow adult mort 5%/10 years	–0.028 \pm 0.060	0.848	3,199 \pm 1,863	74
13	Baseline + fishing mortality 105/year 82♀:18♂ and \uparrow pup mortality 25% and \uparrow adult mort 5%/10 years	–0.039 \pm 0.064	0.982	3,536 \pm 2,226	59

3 and 4 had a bycatch level of 105 NZ sea lions per year (between 1995–1996 and 2007–2008) with the two differing sex ratios. The estimate of 105 is average estimated number of interactions of NZ sea lions and trawlers within SQU6T (both arrow squid and scampi), which would be the number of estimated mortalities if no SLEDS were used in the SQU6T arrow squid fishery (as defined in Thompson et al. 2010). The estimates of Thompson et al. (2010) are currently used by NZ Ministry of Fisheries for the NZ sea lion/squid fisheries management (<http://www.fish.govt.nz>).

PVA simulations were run 500 times (Harris et al. 1987; Miller and Lacy 2005). The time frame for simulations was 100 years with ‘quasi-extinction’ defined as the population reducing to less than 1,000 animals. Quasi-extinction at 1,000 was chosen because at low population numbers, extinction of any species can quickly occur and many species develop problems such as loss of genetic variability or mortality before breeding (Pimm 1991). This is recognised in vertebrate populations as there are species numbering in the low thousands of breeding pairs that are listed as critically endangered and have little security in population recovery (Mace 1996; IUCN 2008). Traill et al. (2007) recommended for mammals that minimum viable population size would be 3,876 (95% CI 2,261–5,095); therefore, setting quasi-extinction at 1,000 is set lower than recommend for long-term species survival and would be the equivalent to functional extinction.

Density dependence was excluded from the models because density dependence is not evident in the NZ sea lion population dynamics (Breen et al. 2003; Breen and Kim 2006; Breen et al. 2010; Chilvers and Mackenzie 2010; Chilvers et al. 2010). The population has also been decreasing rapidly in the last decade with no apparent change in population parameters that would indicate density dependence (Chilvers and Mackenzie 2010; Chilvers et al. 2010). Added to these observations, Ginzburg et al. (1990) suggested that when density dependence is unknown or available data sets are insufficient for reconstructing reliable measurements of density dependence, estimates of extinction probabilities are better made from models omitting density dependence.

There are currently limited data available on the population genetics of NZ sea lions; therefore, this was also excluded from the model. However, data that are available on microsatellite diversity indicate that NZ sea lions do not have lower than expected heterozygosity or low average number of alleles per locus relative to other Otariid species (Robertson and Chilvers 2011). These two factors indicate that inbreeding or strong genetic drift in the current population is unlikely, however does not mean that it could not suffer in the future if numbers decline further.

Model fit testing

The above models as were rerun 500 times over a 15-year period (1995–2010, otherwise using the same criteria) to

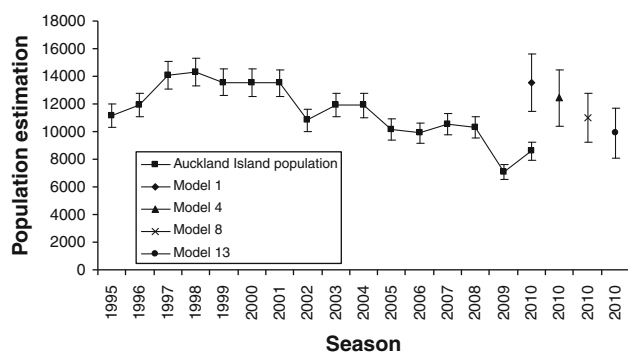


Fig. 2 Auckland Island field data population estimation 1994/1995–2009/2010 with the mean modelled population size at 2010 (\pm SD) for models 1, 4, 8 and 13

enable comparison with empirical field data from the Auckland Islands during the same period. As one of the main predictive outputs for comparison of the models is population size, the pup production estimates from the Auckland Islands from this period (Chilvers et al. 2007; Chilvers 2011) were scaled up into population estimates using equations from Gales and Fletcher (1999, Fig. 2).

Results

PVA models

Table 2 shows the predictions of models 1–13. The basic model, with no fisheries bycatch deaths and no bacterial epizootics, shows that the Auckland Island population of NZ sea lions would increase over the next 100 years and could reach a population size of over 25,590 (\pm 9,407). When the current naturally occurring bacterial epizootics are added to the basic model, there is still no likelihood of quasi-extinction in any of the three models; however, bacterial epizootics affecting adult mortality by 5% show a static population size over 100 years (Model 4, Table 2). When pup survival only is decreased due to bacterial epizootics, even at once every 5-year occurrence intervals, the Auckland Island population would still increase over 100 years,

but at a reduced rate to the baseline model (Models 2 and 3, Table 2).

With any level of fishing mortality and varied sex ratios (models 5–8), the Auckland Island population decreases, particularly when the higher proportion of females are affected by bycatch (Table 2). Similarly, when the combination of fishing mortality (89/year various sex ratios) and bacterial epizootics occur (models 9–12), all models show the Auckland Island NZ sea lion population declining. Model 13, with a bycatch of 105 animals/year with a sex ratio of 82♀:18♂ since 1995 and bacterial epizootics that effect both pup and adult survival once every 10 years, shows that the Auckland Island population has a quasi-extinction probability of 0.98 in 59 years.

Model fit testing

Figure 2 represents the estimated population decline from field data for the NZ sea lion population at the Auckland Islands from 1995 to 2010 based on pup production estimates (Chilvers et al. 2007; Chilvers 2011) modelled to population estimates using Gales and Fletcher (1999). The population has declined ($r = 0.026$) over the 15-year period. The predicted mean population size after 15 years for Models 1 (baseline), 4 (highest impact for bacterial catastrophe), 8 (highest bycatch level and most distorted sex ratio) and 13 (highest level of impacts from both factors) is shown in Fig. 2 (\pm SD, Table 3). Mean annual rate of population change across the 15-year simulations ($r \pm$ SD) for these same models is given in Table 2. The only model for which the predicted population size estimate and standard deviation overlap with the empirical field data for the NZSL Auckland Islands population is Model 13.

Discussion

The pup production of the Auckland Island population of NZ sea lions has been in decline since 1997/1998 (Chilvers et al. 2007; Chilvers 2008; Fig. 2). Pup production declined

Table 3 Predictions of models 1, 4, 8 and 13 for Auckland Island New Zealand sea lions over the 15 years, 1995–2010, include mean annual rate of population change across the simulations ($r \pm$ SD) and the mean modelled population size in 2010 (\pm SD)

Model	Description	Model mean population change (r) (1995–2010)	Model mean population size by 2010
1	Baseline	0.012 \pm 0.051	13,547 \pm 2,070
4	Baseline + pup mortality \uparrow 25% and adult mortality \uparrow 5% once every 10 years	0.006 \pm 0.054	12,424 \pm 2,013
8	Baseline + fishing mortality 105 82♀:18♂ ratio	−0.002 \pm 0.051	10,973 \pm 1,759
13	Baseline + fishing mortality 105/year 82♀:18♂ and \uparrow pup mortality 25% and \uparrow adult mort 5%/10 years	−0.009 \pm 0.055	9,885 \pm 1,840

by over 50% between 1997/1998 and 2008/2009 and has been linked to decreasing numbers of adult females (Chilvers 2008; Baker et al. 2010). The results of this simple PVA indicate that the Auckland Islands NZ sea lion population should be at least stable, if not increasing, even when taking into account the naturally occurring epizootic events (Table 2, Models 1–4). However the population is not (Fig. 2). Results indicate that current fishing mortality could result in population decline. Although none of the bacterial epizootic models (Model 2–4) put the population into decline, they all show the populations growth rate decline, reducing the populations ability to sustain any other forms of mortality. If the worst bacterial epizootic (1998) was to occur once every 10 years, the population could not sustain any fishing mortality.

It is rare to have 15 years of empirical field data to be able to test against model predictions. The only model for which the predicted population size estimate and standard deviation overlap with the empirical field data for the NZSL Auckland Islands population over the last 15 years is Model 13. Model 13 is the ‘severest’ of models presented here with the highest estimated level of fisheries bycatch and greatest sex ratio difference (105 animals per year with skewed sex ratio towards females 82:18) and the highest level of bacterial impact including adult deaths. Yet, this model only just overlaps with the actual population change seen in the Auckland Island NZ sea lion population. This indicates that there may be even greater levels of mortality or unknown factors impacting on this population.

The fisheries’ mortality used in the models is optimistic in their predictions because (1) the fisheries bycatch mortality modelled does not take into account any other fishing mortality occurring near the Auckland Islands. In addition to SQU6T mortality, an average of 11 animals a year is estimated incidentally killed for other fisheries in the sub-Antarctic area, particularly southern blue whiting (*Microsistius australis*) fisheries around Campbell Island and fisheries just north of the Auckland Island shelf on the Snares shelf (Thompson et al. 2010); (2) the modelling does not account for other forms of anthropogenic mortality on this NZ sea lion population outside the Auckland Island or sub-Antarctic area, such as potential mortality from other fisheries around Stewart Island or NZ mainland where juveniles and adult males migrate to out of the breeding season (Robertson et al. 2006); and (3) it does not account for indirect impacts (which may act in a variety of ways including mortality, reduced productivity or injury), such as resource competition and habitat disturbance cause by the fishing or other anthropogenic influences.

This PVA does not account for the entire NZ sea lion population, i.e. 29% of pups born in 2009/2010 were on Campbell Island (Maloney et al. *in press*). However, there are no accurate population parameter estimates for the

Campbell Island population (i.e. survival or reproductive rates) except recent pup production estimates in 2002/2003 (385), 2007/2008 (583) and 2009/2010 (681) (Childerhouse et al. 2005; Maloney et al. 2009, *in press*; respectively). Due to this limited population parameter availability, it was considered that the pup production figures from Campbell Island could not just be added into this model with the Auckland Island population parameters. It should be noted, however, that the pup production estimates at Campbell Island indicate that pup production is increasing. There is significantly less fishing activities and fewer numbers of animals caught around the Campbell Island area, with no arrow squid trawl fishing and fishing activity predominantly further from shore and catching juvenile males (Thompson et al. 2010). Therefore, the increasing pup production at Campbell Island, where there is significantly less fishing activity and bycatch, but where bacterial epizootics are known to have impacted, supports this research that when fisheries’ pressures are high and bycatch is predominantly female, it appears to significantly affect population decline.

Population viability analysis techniques are not often applied to entire pinniped species, due to the large population size typical of most species. They are, however, very appropriate for small and isolated populations such as NZ sea lions and have been used on other similar pinniped species, e.g. harbour seals, *Phoca vitulina* (Bjoerge et al. 1994), Saimaa ringed seals, *Phoca hispida* (Kokko et al. 1998) and southern elephant seals, *Mirounga leonine*, at the Falkland Islands (Gamlimberti et al. 2001). Two criticisms of PVA analysis is that there is often too few data to run an appropriate analysis and that factors are omitted that are unknown but that could affect PVA analysis, e.g. possible genetics impacts on a population, density dependence effects on populations or effects of catastrophes. The first criticism of lack of data is not relevant to the Auckland Island NZ sea lions as the population has been monitored and data on population parameters determined and widely published. Lack of data for the Campbell Island population is relevant and is an area where more research is needed. Genetic impacts on the population and density dependence affects on the population were not included in this model as explain in the methods. The known ‘catastrophes’ of bacterial epizootics have been included within this analysis; however, other possible catastrophes such as oil spills, tsunamis or other unexpected catastrophes are possible for this species and have not been included in this analysis and therefore should be considered when interpreting results and putting in place appropriate management. Environmental stochasticity is including in the models through the use of environmental variability (EV) for all survival and reproductive rates (Table 1). This EV is apparent in confidence intervals around the model estimated population means and rate of population change (Table 2).

Management implications

There have been repeated concerns raised over the decreases in pup production seen for NZ sea lions, the least abundant sea lion species in the world with a severely restricted number and distribution of breeding areas (Wilkinson et al. 2003; Chilvers 2008; Baker et al. 2010; Robertson and Chilvers 2011). The current rate of decline in the Auckland Island population would result in this population being functionally extinct by 2035, 24 years from now. In accordance with the New Zealand Marine Mammals Protection Act 1978, NZ sea lions as a threatened species have to ‘...be managed to a level of human-induced mortality which would allow the species to achieve non-threatened status as soon as reasonably practicable, and in any event within 20 years’. This modelling indicates that the current level of bycatch from the trawl fisheries within the SQU6T area is the most significant known negative impact on the population, and therefore, the aim of management, as highlighted in the NZ Department of Conservation NZ sea lion species management plan 2009–2014 (<http://www.doc.govt.nz/upload/documents/science-and-technical/sap251entire.pdf>), must be to reduce anthropogenic mortality and impacts on this species, particularly fisheries bycatch deaths. If the naturally occurring bacterial epizootics continue to occur, bycatch mortality from fisheries needs to be zero for the Auckland island NZ seal lion population.

One of the greatest difficulties for conservation and management is when there are dual goals of sustaining economic exploitation activities, such as fisheries, and protecting wildlife. The Auckland Islands are an area where these joint goals could be achieved by changing at least the squid fishing technique from trawling to jigging. An example of this method of fishing being conducted in a similar environment is the offshore squid jig fishery at the Falkland Islands (Barton 2002). Such a change in fishing method could reduce the fishing-related mortality of NZ sea lions (and other wildlife) to the needed zero or negligible level (Arnould et al. 2003) while allowing the continuation of squid fishing.

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