

Current bycatch levels in Auckland Islands trawl fisheries unlikely to be driving New Zealand sea lion (*Phocarctos hookeri*) population decline

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ABSTRACT

1. New Zealand sea lions are incidentally killed in trawl fisheries around the Auckland Islands with most mortality having been attributed to the Auckland Islands squid fishery. Fishery management measures include the establishment of a 12 nautical mile marine reserve around the Auckland Islands excluding all fishing within that range, the instigation of mortality limits that can trigger spatio-temporal closures, and widespread use of a 'Sea Lion Excluder Device' (SLED) that allows sea lions to escape from a trawl net. Although there has been controversy regarding SLED efficacy, the evidence from numerous research trials and assessments is that SLEDs have contributed to reduced rates of sea lion bycatch in the Auckland Islands squid fishery.

2. Population viability analysis (PVA) modelling, using VORTEX, of the Auckland Islands New Zealand sea lion population was undertaken to ascertain if the reported levels of bycatch of sea lions in trawl fisheries around the Auckland Islands are sustainable following substantial and effective mitigation to reduce bycatch, particularly in the Auckland Islands squid fishery.

3. Modelling indicated slow population growth of the Auckland Islands New Zealand sea lion population with current bycatch estimates from all Auckland Islands trawl fisheries. Additional modelling seeking explanations for observed population declines over the last decade indicated that epizootic events that reduce pup production may have a greater impact on population growth, especially if these events are more frequent than previously assumed.

4. Modelling results suggest that sea lion bycatch in the squid fishery and other trawl fisheries around the Auckland Islands is unlikely to be currently having a significant impact on the Auckland Islands New Zealand sea lion population. Therefore, resources should be directed towards other hypotheses for any continued sea lion population decline as well as continued refinement of mitigation techniques to reduce fisheries-related mortality.

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Received 13 February 2014; Revised 13 July 2014; Accepted 16 August 2014

KEY WORDS: ocean; endangered species; modelling; sustainability; mammals; fishing; trawling

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INTRODUCTION

The New Zealand sea lion (hereafter referred to as 'sea lion'), *Phocarctos hookeri*, a New Zealand endemic, was classified as Vulnerable on the IUCN Red List in 2008 based on a 30% decline in pup production at some of the major breeding colonies in the preceding 10 years (Gales, 2008; Figure 1). It has a Nationally Critical status on the New Zealand Threat Classification system (Baker *et al.*, 2010). The species once occurred from the northernmost cape of New Zealand's North Island to sub-Antarctic Campbell Island (Childerhouse and Gales, 1998). However, historically, subsistence hunting followed by commercial sealing greatly reduced both the population and the breeding distribution (Childerhouse *et al.*, 2010a).

The species' birthing season occurs from mid-December to early January each austral summer¹ with breeding male and female sea lions likely to be ashore for prolonged periods between late November and January (MAF (Ministry of Agriculture and Forestry), 2012). Pregnant females give birth to a single pup in late December, stay ashore for about 10 days after giving birth and then alternate between foraging trips and returning to suckle their pups for a further 10–12 months before weaning (MAF (Ministry of Agriculture and Forestry), 2012). Based on pup production estimates from the Auckland Islands in 2009/2010 (Chilvers, 2012a) and Campbell Island in 2007/2008 (Maloney *et al.*, 2009), 76% of all sea lion pups are born at the Auckland Islands with most others born at Campbell Island and limited breeding reported from the Snares Islands, Stewart Island and the Otago Peninsula (MPI (Ministry for Primary Industries), 2012; Figure 2).

Commercial trawl fisheries have been implicated in the observed decline of sea lions owing to the incidental mortality (hereafter referred to as 'bycatch'²) of sea lions in trawl nets (Robertson and

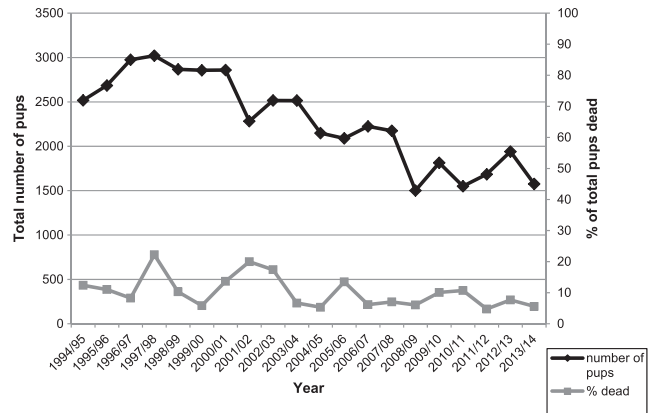


Figure 1. Total estimated pup production and the proportion of those recorded dead during annual mark/recapture estimate field work (late December to mid-February) for New Zealand sea lions at the Auckland Islands 1994/1995–2013/2014. Data before 2012/2013 are from Chilvers (2012a); data for 2012/2013 and 2013/2014 from Childerhouse (2014).

Chilvers, 2011). Most fisheries-related mortality of sea lions has occurred in the Auckland Islands squid fishery (Thompson *et al.*, 2013; Figure 3; Table 1). However, mortality has also been recorded or estimated in the Auckland Islands scampi fishery, Auckland Islands non-squid/scampi trawl fisheries, the southern blue whiting fishery operating near Campbell Island and the Stewart-Snares shelf trawl fisheries (Thompson *et al.*, 2013; Table 1). The sea lions caught in the southern blue whiting trawl fishery are considered to emanate from the Campbell Island population owing to the proximity of this fishery to Campbell Island and because no tagged animals from Auckland Island colonies have been observed killed in the fishery to date (DOC, unpublished data). However, occasionally Auckland Island animals may be affected in this fishery as there is limited evidence that some male sea lions from the Auckland Islands forage around Campbell Island and may overlap spatially with the southern blue whiting trawl fishery near Campbell Island (Geschke and Chilvers, 2009; Maloney *et al.*, 2012). Satellite tracking data also showed one record of a nursing female sea lion from the Auckland Islands undertaking one return trip to Campbell Island (S. Childerhouse, personal communication). Only one sea lion (a female from March 2005) caught in the Stewart-Snares shelf trawl fisheries has had an identifying brand or tag and it originated from

¹For clarity, breeding seasons in this paper are referred to by the two calendar years that they span (e.g. 2001/02 breeding season).

²For the purposes of this paper, the term 'bycatch' is used, in general, to cover the incidental capture of sea lions in trawl nets. It is also assumed that captures are the equivalent to mortalities. Note that in New Zealand literature the term 'bycatch' is used to describe non-target fish catch of commercial value and 'incidental mortality' is used in relation to sea lions killed in fishing nets.

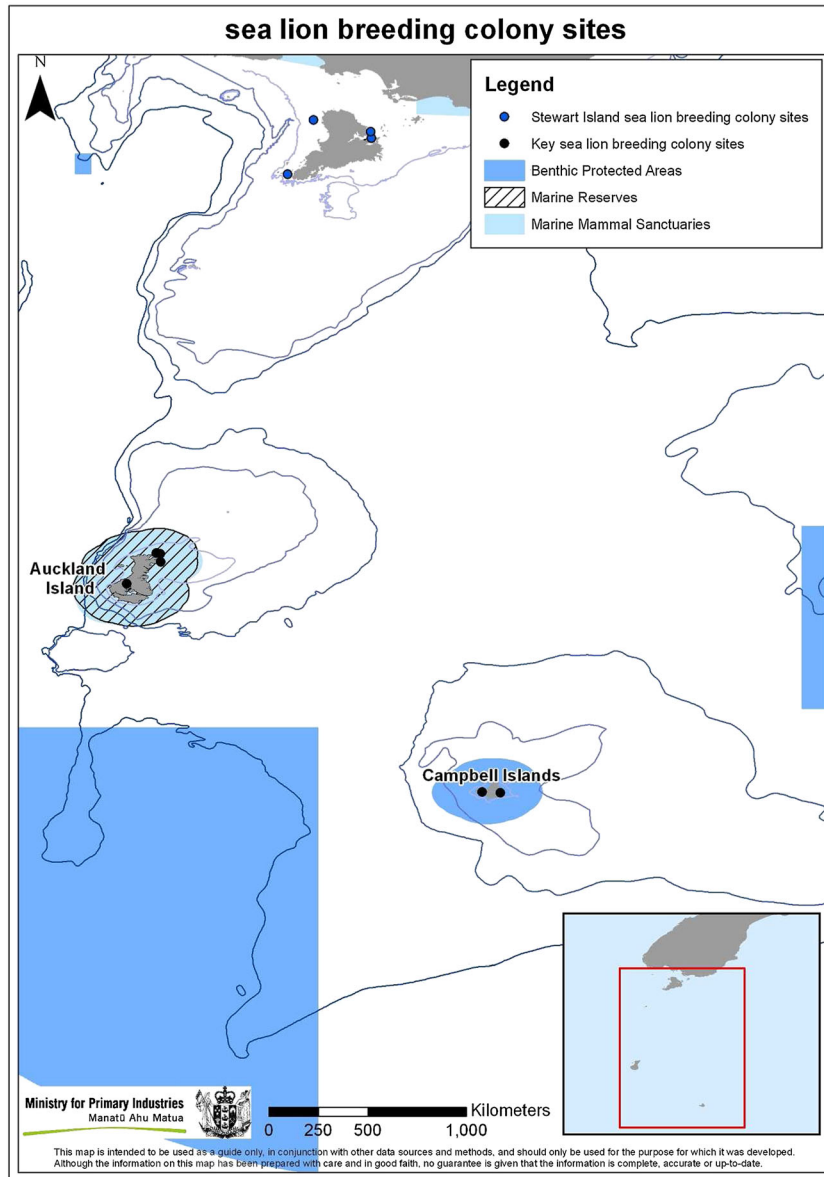


Figure 2. Location of New Zealand sea lion breeding populations at the Auckland Islands and Campbell Islands (map provided by Ministry for Primary Industries, New Zealand).

Sandy Bay, Enderby Island (MPI, unpublished data) although the provenance of other animals caught in this fishery is unknown.

The annual Auckland Islands squid fishery (targeting *Nototodarus sloanii*), one of New Zealand's largest, more valuable fisheries, uses a combination of bottom and mid-water trawls operating at bottom depths of about 150–250 m. Although not completely understood, the foraging areas and depths of sea lions have been shown to overlap, in some areas,

with commercial trawl fishing activity in the Auckland Islands squid fishery (Chilvers, 2008, 2009). Peak activity in the fishery occurs between February and May (MAF (Ministry of Agriculture and Forestry), 2012) coinciding with part of the lactation period for breeding female sea lions. Sea lion mortality in the Auckland Islands squid fishery has been monitored by government observers since 1988 (Wilkinson *et al.*, 2003), although observer coverage has varied from <10% to 99% over this

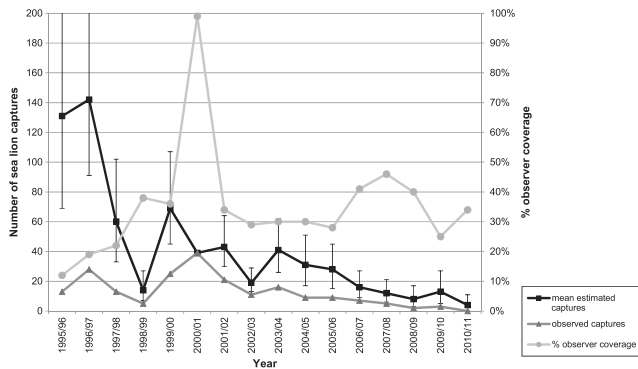


Figure 3. The observed number of captures (dark grey triangles) and mean estimated captures (black squares; error bars = 95% confidence interval) per year of New Zealand sea lions in the Auckland Islands squid fishery from 1995/1996 to 2010/2011. Observer coverage (light grey circles) is also shown. Data from Thompson *et al.* (2013).

period (Figure 3). The mean estimated level of bycatch peaked in the mid- to late-1990s (specifically at 131 in 1995/1996 and at 142 in 1996/1997; Thompson *et al.*, 2013; Figure 3). To reduce the impact of fisheries-related mortality on the sea lion population, management measures have included the establishment of a 12 nautical mile marine reserve around the Auckland Islands

excluding all fishing within that range, the instigation of mortality limits that can trigger spatio-temporal closures, and the development and implementation of a 'Sea Lion Excluder Device' or SLED (MAF (Ministry of Agriculture and Forestry), 2012; Hamilton and Baker, 2014). The SLED comprises an additional section of netting inserted between the lengthener and codend of the trawl net with an angled two or three part metal grid that aims to direct sea lions to an escape hole in the top of the net and exclude them from the trawl codend (Wilkinson *et al.*, 2003; MAF (Ministry of Agriculture and Forestry), 2012; Hamilton and Baker, 2014). Since 2004/2005 all vessels in the Auckland Island squid fishery have used government-specified, standardized SLEDs (MAF (Ministry of Agriculture and Forestry), 2012; MPI (Ministry for Primary Industries), 2012; Hamilton and Baker, 2014). Following widespread SLED use, the annual mean capture estimates of sea lions in the Auckland Islands squid fishery declined from 31 in 2004/05 to four in 2010/11, although there were no observed sea

Table 1. New Zealand sea lion capture data from New Zealand trawl fisheries (data from last five reported seasons, Thompson *et al.*, 2013)

Fishery	Location	Target species	Year	Total annual trawl effort (tows)	Observer coverage (%)	Mean estimated captures (95% CI)
Southern blue whiting trawl fishery (SBW6I)	Campbell Islands	southern blue whiting (<i>Micromesistius australis</i>)	2007	544	32	15 (6–29)
			2008	557	41	8 (5–14)
			2009	627	20	1 (0–7)
			2010	550	43	24 (15–36)
			2011	815	40	15 (8–25)
Stewart-Snares shelf trawl fisheries	southern end of the Stewart-Snares shelf area	primarily squid but also hoki (<i>Macruronus novaezelandiae</i>), jack mackerel (<i>Trachurus</i> spp.) and barracouta (<i>Thyrstites atun</i>)	2006/07	3 498	24	4 (1–7)
			2007/08	3 249	36	3 (1–7)
			2008/09	2 547	31	2 (0–5)
			2009/10	2 784	43	3 (1–6)
			2010/11	2 456	36	1 (0–4)
Auckland Islands scampi trawl fishery	Auckland Islands	scampi (<i>Metanephrops challengeri</i>)	2006/07	1 328	7	10 (4–19)
			2007/08	1 327	7	9 (2–18)
			2008/09	1 457	4	11 (4–21)
			2009/10	940	10	6 (1–13)
			2010/11	1 401	15	9 (2–17)
Auckland Islands non-squid/scampi trawl fishery	Auckland Islands	orange roughy (<i>Hoplostethus atlanticus</i>) and hoki	2006/07	38	5	0 (0–1)
			2007/08	147	45	0 (0–2)
			2008/09	121	50	0 (0–2)
			2009/10	77	66	0 (0–1)
			2010/11	131	37	0 (0–2)
Auckland Islands squid trawl fishery (SQU6T)	Auckland Islands	arrow squid (<i>Nototodarus sloanii</i>)	2006/07	1 320	41	16 (9–27)
			2007/08	1 265	46	12 (6–21)
			2008/09	1 925	40	8 (3–17)
			2009/10	1 190	25	13 (5–27)
			2010/11	1 586	34	4 (0–11)

lion captures in 2010/2011 (mean estimated captures = 4; range 0–11; 34% observer coverage; Thompson *et al.*, 2013; Figure 3) and 2011/2012 (MPI, unpublished data). However, the annual sea lion population estimates (based on estimates of pup numbers) continued to decline to the lowest reported estimate of 1501 pups in 2008/2009 (Figure 1: Chilvers *et al.*, 2007; Robertson and Chilvers, 2011; Chilvers, 2012a; Childerhouse, 2014). This led to uncertainty regarding the efficacy of SLEDs with claims that some animals could suffer head trauma from hitting the SLED's hard grid that may compromise their post-escape survival (Robertson and Chilvers, 2011). Although research on the effectiveness of SLEDs has been complex, concerns surrounding the efficacy of SLEDs in allowing most sea lions to exit a trawl net and survive have been shown to be largely unfounded (Hamilton and Baker, 2014). Nevertheless, the impact of the Auckland Islands squid fishery, in particular, on the sea lion population has continued to be a contentious issue particularly given additional concerns based on an apparent increased female bias in bycatch (Robertson and Chilvers, 2011).

The other fishery currently recording noteworthy levels of interactions with sea lions around the Auckland Islands is the Auckland Islands scampi (*Metanephrops challenger*) fishery, which utilizes light bottom trawl gear operating at 200–500 m on the continental slope (MPI (Ministry for Primary Industries), 2013). This fishery, which does not deploy SLEDs, recorded annual mean capture estimates of 6–11 sea lions for 2006/2007 to 2010/2011 (Thompson *et al.*, 2013; Table 1). The Auckland Islands non-squid/scampi trawl fishery, which primarily targets orange roughy (*Hoplostethus atlanticus*) and hoki (*Macruronus novaezelandiae*), has also recorded low levels of interactions with sea lions in the past although the mean estimated captures since 2004/2005 has been zero (Thompson *et al.*, 2013; Table 1).

There have been concurrent 'natural' impacts on the Auckland Islands sea lion population that have added to the complex nature of understanding the effect of incidental fisheries-based mortality. Epizootic events resulted in the deaths of 53%, 32% and 21% of pups produced for the 1997/1998, 2001/2002 and 2002/2003 seasons (Figure 1),

respectively, with additional adult female mortality also occurring during the 1997/1998 event (Wilkinson *et al.*, 2003; Castinel *et al.*, 2007; Chilvers, 2008; DOC (Department of Conservation), 2009). In 2008/2009, a 31% drop in pup production in one year (to 1501 pups, Figure 1) was attributed to females not returning to breed in that year although the cause of this was not established (Robertson and Chilvers, 2011). Researchers undertake mark-recapture studies of the sea lion population on the Auckland Islands from late December to mid-February every year. However, there is recent evidence that disease events may be occurring on the colony after this time and, therefore, pup mortality from epizootics may be underestimated (S. Childerhouse, personal communication).

To inform Auckland Island squid fishery management, modelling has been undertaken over a number of years to estimate sea lion population projection and evaluate the population consequences of alternative mortality control rules (Breen *et al.*, 2010). Chilvers (2012b) undertook a population viability analysis (PVA) that predicted the 'functional extinction' (i.e. 'quasi-extinction' set at 1000 animals) of the Auckland Islands sea lion population in less than 100 years from 1995 and concluded that the level of bycatch from trawl fisheries around the Auckland Islands was the most significant known negative impact on the sea lion population. However, this modelling did not consider the effectiveness of mitigation, particularly over the last 10 years, that has resulted in fisheries-related mortality being greatly reduced (Thompson *et al.*, 2013; Figure 3). Therefore, the conclusion that fisheries-based mortality continues to be the factor driving sea lion population decline (Chilvers, 2012b) may no longer be valid and other factors may be contributing to the continued sea lion population decline at the Auckland Islands.

Clarification of the current possible causes of the observed sea lion population decline is necessary if effective management actions are to be developed to reverse this decline. For this reason, further assessment of the impact of the Auckland Islands squid fishery, as well as other trawl fisheries around the Auckland Islands, on the viability of the sea lion population, is needed. Taking into account the efficacy of SLEDs and other mitigation measures, the aim of this work was to undertake a PVA of the Auckland Islands sea lion population to understand

the key demographic factors driving trends in the population and enable evaluation of current levels of fisheries-related mortality.

METHODS

A population model was developed based on the most robust population dynamics data for the New Zealand sea lion. To promote transparency, the program VORTEX (Version 10; Lacy and Pollak, 2014) was used as it is freeware that is easily accessed and understood and is widely used for undertaking PVAs in a range of situations (Prowse *et al.*, 2013; Midwood *et al.*, 2014). This program simulates survival and reproductive events in successive years for each individual in a population by the Monte Carlo method. It is stochastic in that it imposes variations in annual survival and reproduction by random number generations according to prescribed probability distributions for reproduction and survival rates.

Parameter values used in the modelling

The PVA modelling was developed using published data for sea lions from a New Zealand Department of Conservation long-term demographic study on the Auckland Islands population. It was assumed that sea lions killed in the Auckland Islands fisheries were from the Auckland Islands breeding colonies. The input demographic parameters are summarized in Table 2 with further information below. Each model was run for a 30 year period with 2000 simulations.

The mortality estimates modelled were based on the inverse of survival estimates published graphically in [Chilvers and MacKenzie \(2010\)](#) (with data subsequently clarified by D. MacKenzie) and modified to account for levels of fishing mortality embedded in them. As these published survival estimates were based on tag resights (for age classes 0, 1, 2, 3, and >4 years) from the Auckland Islands sea lion population from 1998–2005 (Chilvers and MacKenzie, 2010), they intrinsically included existing levels of fisheries mortality as well as mortality from epizootics. Therefore, for age classes 4 years and over (i.e. the age range predominantly killed in fisheries interactions and the non-pup age

Table 2. New Zealand sea lion demographic parameters, incidental fisheries capture levels and mass mortality disease levels used in the Population Viability Assessment (PVA) for the Auckland Islands New Zealand sea lion population. Further details of how parameter values were derived are in Methods

Parameter description	Value(s) modelled
Inbreeding depression	None
CV concordance of reproduction and survival	None
Breeding strategy	Polygynous (Chilvers, 2012b)
Young per year	1
Female breeding age (years)	6 (Chilvers <i>et al.</i> , 2010)
Female maximum breeding age (years)	25 (Childerhouse <i>et al.</i> , 2010b)
Male breeding age (years)	9 (Robertson <i>et al.</i> , 2006)
Male maximum breeding age (years)	25 (S. Childerhouse pers. comm.)
Maximum life span	25
Mean percentage adult females producing progeny/year (EV = environmental variation)	67% (EV = 10) (Childerhouse <i>et al.</i> , 2010b)
Sex ratio at birth (males)	51% males (Chilvers, 2012b)
Density-dependent reproduction	No
Percentage of males in breeding pool	23 (Robertson <i>et al.</i> , 2006)
Female mortality (%)	<4 year mortality data derived from Chilvers and MacKenzie (2010); ≥4 year mortality data from Chilvers (2012b)
Age 0	47 (8)
Age 1	32 (8)
Age 2	20 (6)
Age 3	14 (4)
Age 4	4 (2)
Age 5	4 (2)
Adults	2 (1)
Male mortality (%)	
Age 0	62 (14)
Age 1	34 (8)
Age 2	15 (6)
Age 3	6 (2)
Age 4	2 (1)
Age 5	2 (1)
Age 6	2 (1)
Age 7	2 (1)
Age 8	2 (1)
Adults	2 (1)
Initial population size	12 065 in 2009 (MPI (Ministry for Primary Industries), 2012)
Additional scenario options added to the Base Models:	
Harvest (annual mortality in fisheries)	(i) 68 animals (52 F:16 M) (ii) 20 animals (15 F:5 M) (iii) 50 animals (39 F:11 M)
Catastrophe (disease)	
- Frequency	(i) 6.7% (ii) 25% (iii) 50% (iv) 100%
- Reproduction	0.50
- Survival	1.0

range reported to die in the 1998 bacterial epizootic) the relevant age-class mortality estimates reported in Chilvers (2012b) were used, which had been adjusted to exclude fishing-related and epizootic

mortality (Chilvers, 2012c; L. Chilvers, personal communication). For age classes less than 4 years, mortality estimates were averaged from the Chilvers and MacKenzie (2010) data excluding their estimates for 2004 and 2005 which were based on small sample sizes (Table 2).

Female sea lions are thought to reproduce from 3–26 years of age (based on evidence of lactation) although most do not breed until they are 6 years old and no female older than 25 years has been observed with a pup (Childerhouse *et al.*, 2010a, b; MPI (Ministry for Primary Industries), 2012). The mean observed reproductive rate for females 3–28 years old was 0.67 (SE = 0.01) (Childerhouse *et al.*, 2010b). Therefore, the ‘Age of first offspring for females’ (i.e. age at which the typical female produces her first offspring) was modelled as 6 years, ‘Maximum age of reproduction’ for females as 25 years and ‘Mean % of adult females producing progeny per year’ as 67% (Table 2). It was also assumed that males would continue breeding for at least as long as females (S. Childerhouse, personal communication) and, therefore, ‘Maximum age of reproduction’ for males was also modelled as 25 years (Table 2).

Density dependence was not included in the modelling as there is no evidence for this in reported sea lion population dynamics and, when the Auckland Islands sea lion population decreased over the past decade, there was no apparent change in demographic parameters that would indicate density dependence (Breen *et al.*, 2010; Chilvers and MacKenzie, 2010; Chilvers *et al.*, 2010; Chilvers, 2012b).

Inbreeding depression was not modelled because of the limited data available on the population genetics of sea lions, and the assessment that inbreeding or strong genetic drift in the Auckland Islands sea lion population is unlikely (Robertson and Chilvers, 2011; Chilvers, 2012b). Also, owing to the lack of known movement of breeding females (Maloney *et al.*, 2009; Chilvers, 2012b), no level of dispersal was incorporated into the models.

The impact of epizootics was modelled through the ‘Catastrophe’ option in VORTEX by assuming a disease event that occurred randomly every 15 years (6.7% frequency) killed half the pups born

in that year (Reproduction 0.50) but had no impact on survival of other ages (Survival 1.0). These estimates were based on the highest impact epizootic event recorded at the Auckland Islands in 1997/98 (Chilvers, 2008; DOC (Department of Conservation), 2009). Although the 1997/1998 epizootic also affected the adult population, this impact was not factored into the modelling because the evidence was that this was not a regular occurrence. There is recent evidence that the frequency and magnitude of disease events may have been underestimated and that disease may be affecting numbers of pups later in the season and, therefore, not detected during the research programme (S. Childerhouse, personal communication). Therefore, disease events (affecting half the pup production i.e. Reproduction 0.5) were also modelled at the following hypothetical frequencies: every 4th year (25% frequency), every 2nd year (50% frequency) and every year (100% frequency) (Table 2).

As for Chilvers (2012b), it was assumed that animals caught in the Auckland Islands squid fishery, the Auckland Islands scampi fishery and the Auckland Islands non-squid/scampi trawl fisheries were all from the Auckland Islands sea lion population. Therefore, data from all Auckland Island trawl fisheries were included in the modelling (Table 1). Between 2004/2005 and 2008/2009, after SLEDs were introduced and were being refined, it was reported that females accounted for 71% of the observed number of sea lions captured in the Auckland Islands squid fishery (Robertson and Chilvers, 2011). This led to a claim that this bias towards female mortality may be a major contributing factor towards the continued observed population decline at the Auckland Islands as the mortality of breeding females is likely to have a larger impact than male mortality (Robertson and Chilvers, 2011). The sex ratio of 77 female:23 male applied in the PVA scenarios (Table 2) reflects the observed mean sex ratio of 71 female:29 male scaled to estimated mortalities (MPI (Ministry for Primary Industries), 2012). Using the ‘Harvest’ option in VORTEX, the following different levels of fisheries-related mortality were modelled:

- a) as a conservative approach, a ‘high’ annual loss of 68 adult sea lions (52 females, 16 males) per year was modelled (Table 2) based on the most recent

'Fisheries-Related Mortality Limit' set for 2010/11 (MPI (Ministry for Primary Industries), 2012). Although SLEDs allow sea lions to escape and survive an interaction with a trawl net, concerns over potential levels of undetected fisheries-related mortality have resulted in management criteria in which the interaction between sea lions and the Auckland Islands squid fishery is managed through a Fisheries-Related Mortality Limit. The Fisheries-Related Mortality Limit has two components: a strike rate (the number of sea lions presumed to be killed in the fishery in the absence of SLEDs, currently set at 5.89 sea lion interactions per 100 trawl hauls based on previous observer data) and a SLED discount rate which provides a discount on this strike rate to reflect the increased likelihood that a sea lion that enters a trawl net will exit via the SLED and survive (currently set at 82%; MPI (Ministry for Primary Industries), 2012);

- b) a 'medium' mortality level of 50 adult sea lions (39 females:11 males) was modelled based on the sum (from all Auckland Islands trawl fisheries) of the maximum 95% confidence level of estimated captures for the most recent five seasons of data (Tables 1 and 2; Thompson *et al.*, 2013), and
- c) a lower mortality level of 20 adult sea lions (15 females, 5 males) per year was modelled based on the sum of the averages of the most recent five seasons of annual mean estimated captures for each of the Auckland Islands trawl fisheries (Tables 1 and 2).

Sensitivity analyses

Sensitivity analysis was performed to identify the demographic parameters that had the greatest impact on the predictions for sea lion population growth rate and mean final population size following a projected 30 years of modelling.

RESULTS

The Base Model (with no fisheries-based or epizootic mortality applied) showed sea lion population growth at the Auckland Islands of $r=0.015$ and a mean final population size of 19 099 individuals ($SD=3\ 989$) after 30 years of modelling (Model 1, Table 3).

Base model with varying levels of fisheries-related mortality

The scenarios with fisheries-related mortality but no epizootic events applied, showed that

- with an annual mortality reflecting the most recent five years of reported mortality estimates from all Auckland Islands fisheries of 20 (based on annual mean estimated captures) or 50 adult sea lions (based on maximum 95% confidence level of estimated captures), there was no change in the population growth rate ($r=0.015$, Model 2 and Model 3, respectively, Table 3) compared with the Base Model which had no fisheries-related or epizootic mortality; and,
- at the current Fisheries-Related Mortality Limit set at 68 sea lions per year, there was only a slight decrease in the population growth rate ($r=0.014$, Model 4, Table 3) compared with the Base Model.

Base model with varying frequency of epizootic mortality

The scenarios with epizootic events but no fisheries-based mortality showed that, with disease events that killed 50% of annual pup production

- every 15 years, there was a slight decrease in the population growth rate ($r=0.012$, Model 5, Table 3) compared with the Base Model which had no fisheries-based or epizootic mortality;
- every 4 years, the population growth rate ($r=0.006$, Model 6, Table 3) was less than half the Base Model rate;
- every 2 years, there was a negative population growth rate ($r=-0.006$, Model 7, Table 3); and
- every year, showed a negative population growth rate ($r=-0.031$, Model 8, Table 3) resulting in a mean final population size, after 30 years of modelling, that was about one-third ($N=4\ 820$, $SD=921$, Table 3) its original size ($N=12\ 065$).

Base model with high frequency of epizootic mortality and varying levels of fisheries-based mortality

The scenarios with annual epizootic events and additional fisheries-related mortality showed that, with annual mortality of 20 adult sea lions (based on last 5 years of reported annual

NEW ZEALAND SEA LION MORTALITY IN TRAWL FISHERIES

 Table 3. Model predictions for the Auckland Islands New Zealand sea lion population showing mean stochastic population growth rate (Stochastic r), mean final population size (N) and standard deviation (SD) after 30 years. Each model was run for a 30 year period with 2000 simulations

Model	Description	Model explanation	Mean population change	Mean final population size	
			r	N	SD
1	Base Model	No fisheries-based or epizootic mortality	0.015	19 099	3 989
2	Base Model + fishing mortality 20/year at 15 F:5 M ratio	Low level fisheries-based mortality; no epizootic mortality	0.015	19 071	4 069
3	Base Model + fishing mortality 50/year at 39 F:11 M ratio	Medium level fisheries-based mortality; no epizootic mortality	0.015	19 007	3 773
4	Base Model + fishing mortality 68/year at 52 F:16 M ratio	High level of fisheries-based mortality; no epizootic mortality	0.014	18 992	3 854
5	Base Model + disease 0.5 impact on reproduction at 6.7% (every 15 years)	No fisheries-based mortality; epizootic mortality affecting pups every 15 years;	0.012	17 777	3 725
6	Base Model + disease 0.5 impact on reproduction at 25% (every 4 years)	No fisheries-based mortality; epizootic mortality affecting pups every 4 years	0.006	14 585	3 170
7	Base Model + disease 0.5 impact on reproduction at 50% (every 2 years)	No fisheries-based mortality; epizootic mortality affecting pups every 2 years	-0.006	10 446	2 416
8	Base Model + disease 0.5 impact on reproduction at 100% (every year)	No fisheries-based mortality; epizootic mortality affecting pups every year	-0.031	4 820	911
9	Base Model + fishing mortality 20/year at 15 F:5 M ratio + disease 0.5 impact on reproduction at 100% frequency (every year)	Low level fisheries-based mortality; epizootic mortality every year	-0.031	4 829	884
10	Base Model + fishing mortality 68/year at 52 F:16 M ratio + disease 0.5 impact on reproduction at 100% frequency (every year)	High level fisheries-based mortality; epizootic mortality every year	-0.031	4 795	876
Increasing % annual mortality of adult females					
11	Base Model with adult female mortality = 3%	No fisheries-based or epizootic mortality	0.009	16 004	3 069
12	Base Model with adult female mortality = 3% + fishing mortality 20/year at 15 F:5 M ratio	Adult female mortality = 3%; low level fisheries-based mortality; no epizootic mortality	0.009	16 012	3 228
13	Base Model with adult female mortality = 3% + fishing mortality 68/year at 52 F:16 M ratio	Adult female mortality = 3%; high level of fisheries-based mortality; no epizootic mortality	0.008	15 890	3 292
14	Base Model with adult female mortality = 4%	No fisheries-based or epizootic mortality	0.003	13 445	2 731
15	Base Model with adult female mortality = 4% + fishing mortality 20/year at 15 F:5 M ratio	Adult female mortality = 4%; low level fisheries-based mortality; no epizootic mortality	0.003	13 363	2 839
16	Base Model with adult female mortality = 4% + fishing mortality 68/year at 52 F:16 M ratio	Adult female mortality = 4%; high level of fisheries-based mortality; no epizootic mortality	0.002	13 284	2 875

mean estimated captures from all Auckland Islands fisheries) or 68 adult sea lions (current Fisheries-Related Mortality Limit), there was a negative population growth rate of $r = -0.031$ (Model 9 and Model 10, respectively, Table 3). This population growth rate was equivalent to the scenario with the same frequency of epizootic events but with no fisheries-related mortality (i.e. Model 8, Table 3).

Sensitivity analyses

Sensitivity analyses showed that, in particular, the modelled population was sensitive to annual adult female survival and the proportion of females participating in breeding each year. Whilst female adult survival remained greater or equal to 96%, even with 'high' levels of fishing mortality applied, the population continued to grow (Table 3 Models

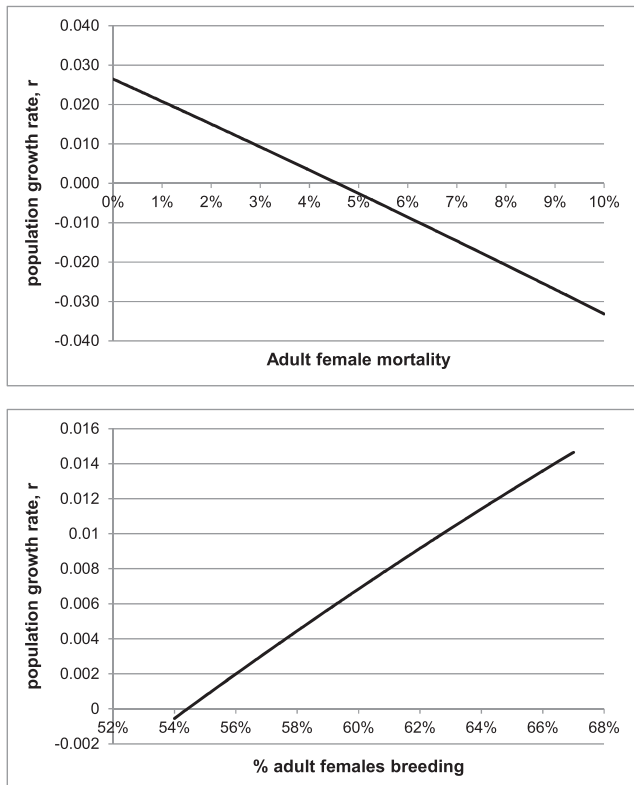


Figure 4. Model sensitivity analyses showing the change in New Zealand sea lion population growth rate given different estimates of adult female annual mortality (top graph) and % female breeding participation (bottom graph). Sensitivity was undertaken using all other parameter values from the Base Model (Table 3, Model 1).

11–16; Figure 4). However, with no fisheries mortality applied, adult female survival below 95% resulted in negative population growth (Figure 4). In the absence of any fishing mortality, the modelled population continued to grow while female breeding participation remained above 55% (Figure 4).

DISCUSSION

New Zealand sea lions from the Auckland Islands population are incidentally killed in trawl fisheries with most of the bycatch having occurred in the Auckland Islands squid fishery. Mitigation management aiming to reduce sea lion bycatch has included the establishment of a 12 nautical mile marine reserve around the Auckland Islands in which all fishing is excluded, the instigation of mortality limits that can trigger spatio-temporal closures, and the design refinement and deployment

of SLEDs on all vessels in the Auckland Islands squid fishery since 2004/2005 (MAF (Ministry of Agriculture and Forestry), 2012). Subsequently, there has been an encouraging reduction in the observed and estimated sea lion bycatch in the Auckland Islands squid fishery (Figure 3; Thompson *et al.*, 2013). Any sea lion that is caught in a trawl net without a SLED will die by drowning whereas correctly deployed SLEDs provide an opportunity for sea lions to escape trawl nets. Extensive efforts to test the efficacy of SLEDs in reducing bycatch have shown that most sea lions are likely to survive following their escape from a trawl net via a SLED (Hamilton and Baker, 2014). To date, SLEDs have not been deployed in any other trawl fisheries around the Auckland Islands because of industry concerns about catch loss (Richard Wells, personal communication). However, if sea lion bycatch continues to be reported at the current (or higher) level in the Auckland Islands scampi fishery (Table 1), it may be worth investigating the feasibility of deploying SLEDs in this trawl fishery.

This paper presents new PVA modelling of the sea lion population at the Auckland Islands. PVA is useful for guiding conservation management and research by identifying the key demographic parameters and impacts that may be affecting the survival of a species. The demographic factors driving trends in the sea lion population were assessed including the impact of different levels of fisheries mortality and epizootic events. The modelling indicated that, even in the absence of both incidental fisheries-related mortality and epizootics, the population growth rate was low ($r=0.015$, Model 1, Table 3). With no epizootic events incorporated but with the addition of fisheries-related mortality levels reflecting recent levels of reported bycatch (Table 1; Thompson *et al.*, 2013), there was no change in the modelled population growth rate (Models 2 and 3, Table 3). There was very little change in the population growth rate even with a relatively high bycatch level of 68 sea lions per year (with 77% female; $r=0.014$, Model 4, Table 3). This indicated that current levels of sea lion bycatch from Auckland Islands trawl fisheries are sustainable, particularly now that effective bycatch mitigation is in place in the Auckland Islands squid fishery. At current

levels, fisheries bycatch is unlikely to be the key factor that is driving population decline for this species.

Modelling epizootic events that killed 50% of annual pup production at varying frequencies had a much larger impact on the population growth rate than applying bycatch levels. With no fisheries bycatch but with a rate and impact level of epizootic event based on the largest episode recorded in the last 15 years, the population growth rate fell to 0.012 (Model 5, Table 3). However, there are recent indications that both the frequency and magnitude of epizootic events may have been underestimated (S. Childerhouse, personal communication). Scenarios modelling more frequent epizootic events in the absence of bycatch effects had a dramatic impact on population growth rates (e.g. epizootics every 4 years, $r = -0.006$, Model 7; every 2 years, $r = -0.006$, Model 7; every year, $r = -0.031$, Model 8; Table 3). Epizootic events affecting pup survival (i.e. 50% reduction in pup production) on an annual basis had the potential to decrease the Auckland Island sea lion population by about 60% over a 30 year time span (Model 8, Table 3). The population trajectory from this hypothetical scenario is similar to the observed population decline over the last 20 years (Figure 1). Modelling both low (20/year, Model 9) and high (68/year, Model 10) levels of bycatch in addition to the high frequency (every year) of epizootic events showed no change in the population growth rate compared with the scenario with no bycatch and high frequency epizootic events. This adds weight to the above conclusion that the current level of bycatch from Auckland Island trawl fisheries is not driving sea lion population decline.

Given the conservative approach taken with the modelling (e.g. modelling high female bias in bycatch and high bycatch rates) and even with the possibility that a small number of animals could be incidentally caught in other trawl fisheries away from the Auckland Islands, this conclusion is especially likely. The mortality of adult female sea lions is likely to have a larger population impact compared with male mortality. It has been suggested that an apparent increasing female bias in mortality estimates for the Auckland Island squid fishery may be contributing to sea lion

population decline (Robertson and Chilvers, 2011). Before effective SLEDs were used in the fishery, it was reported that females accounted for 71% of the observed number of sea lions captured (Robertson and Chilvers, 2011). However, following SLED refinement, observed capture levels declined to less than 10 individuals a year (Thompson *et al.*, 2013). Hence, any perceived impact of a skewed sex ratio is now unlikely to be significant. Therefore, modelling 77% female bias in capture animals for all Auckland Island trawl fisheries provides a conservative modelling approach, especially when applied to the highest modelled mortality of 68 animals per year. It should also be noted that this high level does not reflect current mortality data but is a test of the Fisheries-Related Mortality Limit calculated by the government fisheries regulator (MPI (Ministry for Primary Industries), 2012). Modelling indicated slow population growth when the estimated bycatch levels from all the Auckland Islands trawl fisheries for the past five reported seasons were applied (i.e. 20 or 50 adults, Thompson *et al.*, 2013). However, in the most recent seasons, the reported rates of mortality have been lower than these values. For example, in 2010/2011, for Auckland Islands squid, scampi and non-squid/scampi fisheries combined, the mean estimated mortality was 13 animals (Table 1; Thompson *et al.*, 2013).

Sensitivity analysis can help measure the relative influences of different demographic parameters on population predictions. In particular, the modelling of the Auckland Islands sea lion population was sensitive to changes in adult female annual mortality and the proportion of adult females that breed each year. After adjusting each parameter separately, and in the absence of bycatch or epizootic mortality, population growth rate was negative once adult female mortality fell below 5%, and also when the proportion of females breeding each year fell below 55% (Figure 4), indicating that the continued New Zealand sea lion population decline at the Auckland Islands may perhaps be related to factors affecting decreased breeding productivity.

These PVA modelling results are consistent with other recent population modelling which showed poor correlations between survival of juveniles (ages 2–5 years) and adults (6–14 years) and

fishery-related mortality in the Auckland Islands squid fishery indicating that variation in vulnerable age classes was not primarily driven by the direct effects of fishing (Roberts and Doonan, 2014). With continued commitment to mitigation and maintaining adequate observer coverage in the trawl fisheries around the Auckland Islands, further reductions in estimated mortality levels should be achievable. If direct impacts of fishing operations are no longer a significant problem, resources should be directed towards determining other hypotheses to explain any further sea lion population decline. From the indicative modelling presented here, the severity and frequency of epizootic events and their effect on annual pup production provides a more plausible explanation for the New Zealand sea lion population decline observed at the Auckland Islands in recent years.

ACKNOWLEDGEMENTS

Thanks to the Deepwater Group Ltd (Richard Wells and George Clement) who provided funding for this work and comments on earlier drafts of the manuscript. Daryl MacKenzie and Louise Chilvers provided clarification on estimated survival analyses from their 2010 study. Thanks to Simon Childerhouse and Aleks Terauds who provided comments which greatly enhanced early drafts of this paper. Thanks also to the two anonymous reviewers whose review comments significantly improved this paper.

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