Fisheries New Zealand

Tini a Tangaroa

Population effects of New Zealand sea lion mortality scenarios relating to the southern arrow squid fishery at the Auckland Islands

New Zealand Government

TABLE OF CONTENTS

EXECUTIVE SUMMARY

Roberts, J. (2019). The population effects of New Zealand sea lion mortality scenarios relating to the southern arrow squid fishery at the Auckland Islands.

New Zealand Aquatic Environment and Biodiversity Report No. 223. **35 p.**

A new demographic population model was developed to estimate a revised Population Sustainability Threshold (PST) for New Zealand sea lions in the commercial trawl fishery for Southern arrow squid at the Auckland Islands (SQU 6T). This population model was then used to simulate the projected population effects of alternative levels of future deaths relating to this fishery, including at the PST level.

The female population model was a modification of the previous model used to estimate the PST for this population (Roberts 2017), updated with new pup production estimates and mark resighting observations up to the 2018/19 field season. The model was structured to estimate demographic rates separately for periods of increasing (1990 to 1999), decreasing (1999 to 2009) and stable pup production trajectory (2010 to 2019).

Models also incorporated estimates of southern arrow squid trawl fishery related-deaths, derived from the outputs of separate research projects estimating historical fishery captures (Large et al. 2019) and cryptic mortality levels relating to the use of sea lion exclusion devices (SLEDs) (Meyer 2019).

The base case model produced good fits to pup census, mark-resighting and age distribution observations. Model estimates indicate that the change in pup production trajectory (from decreasing to stable) was driven by increased annual survival at age groups 2–5 and 6–14 years, and a slightly increased annual pupping rate. The model estimate of first year 'pup' survival for the period of stability was virtually unchanged from the period of decline. The base model estimated a current population size of 4293 females in 2019 (95% credible interval (95% CI) = 4120–4473), which was subsequently used in the estimation of the PST.

A new PST criterion was defined by Fisheries New Zealand, using the Spatially-Explicit Fisheries Risk Assessment (SEFRA) approach. A calibration coefficient (ϕ) of 0.1 was selected as the base case value by Fisheries New Zealand, consistent with population recovery to (or preventing depletion below) 95% of unimpacted levels, with future deaths at the PST level. At this level of ϕ, a female-only PST of 26 individuals was estimated. This compares with a mean of 2.8 or 5.4 female deaths in the squid fishery from 2013 to 2017, based on the median or upper 95% CI of estimated deaths for the Auckland Islands southern arrow squid trawl fishery.

Future mortality at the PST (ϕ of 0.1) was estimated to result in a mature female population size in 2025 of between 95.0% (95% CI = 94.7–95.2%) and 96.1% (95% CI = 95.8–96.3%) of what would have occurred in the absence of fishery mortality (depending on the future population growth scenario). Under the stable population growth scenario, future deaths consistent with recent estimated levels would result in a population size in 2025 of between 99.0% (95% CI = 99.0–99.1%) and 99.5% (95% CI = 99.5–99.5%) of unimpacted levels, depending on whether the upper 95% CI, or median of historical deaths was assumed, respectively.

Note that this modelling framework does not account for density dependence, such that the estimates of future population status produced here will be lower than those of comparable assessment models, under which productivity increases as the population declines in size.

1. INTRODUCTION

The New Zealand sea lion has an extremely concentrated breeding distribution with 97–98% of annual pup production at either the Auckland Islands and Campbell Island, in the New Zealand Sub-Antarctic region (Weir et al. 2019, Fyfe 2019, Chilvers 2019, DOC 2019) (Figure 1). The largest breeding population at the Auckland Islands was estimated to have declined by about 40% between the late-1990s and the 2008/09 field season (seasons hereafter referred to by the end year, i.e. 2009), although annual pup production estimates appear to have stabilised since then (DOC 2019; Roberts 2017). The apparent stabilisation of the Auckland Islands breeding population (and increasing populations elsewhere) led to an improvement in the species' New Zealand Threat Classification to "Nationally Vulnerable", the third highest domestic threat rating (Baker et al. 2019). Known threats to the Auckland Islands population include deaths relating to commercial fisheries at the Auckland Islands—including southern arrow squid and scampi trawl fisheries (SQU 6T and SCI 6A, respectively)—*Klebsiella pneumonia* infection-related mortality of pups, and likely indications of limited prey availability and associated nutritional stress (Augé 2010; Meyer et al. 2015; Roe et al. 2015; Roberts & Doonan 2016; Roberts et al. 2018; Stewart-Sinclair 2013).
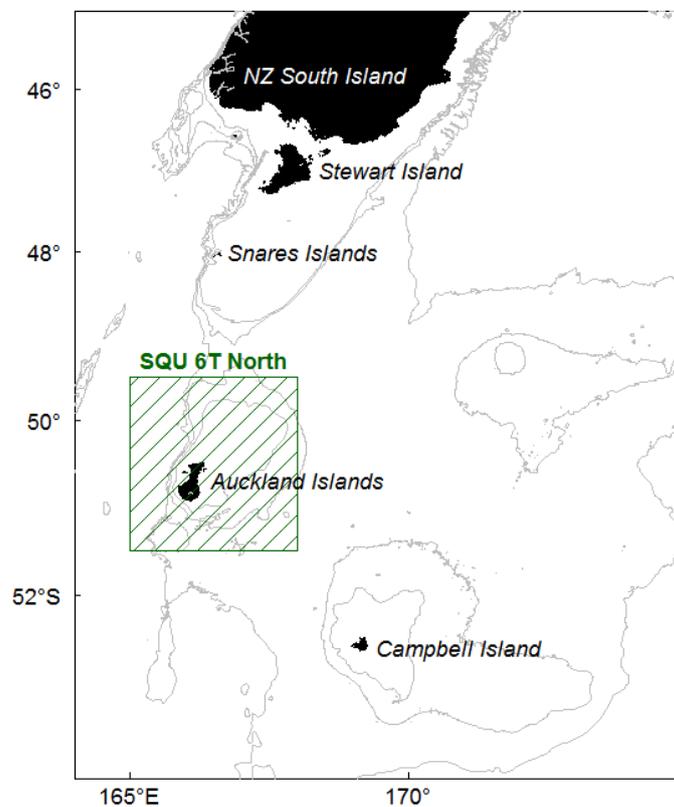
Figure 1: Location of New Zealand sea lion breeding populations, including the Auckland Islands population addressed by this assessment. Grey lines represent the 200 m, 500 m and 1000 m bathymetric contours; the Northern Box of SQU 6T, where the Auckland Islands southern arrow fishery operates, is highlighted with a green hatched polygon.

Previous assessments have estimated mortality thresholds for New Zealand sea lions relating to commercial squid and scampi trawl fisheries at the Auckland Islands (e.g., Abraham et al. 2017; Fletcher 2004; Roberts 2017). Population models have also been developed for New Zealand sea lions at the Auckland Islands to assess the population consequences of fishery-related deaths at different levels (e.g., Breen et al. 2016; Roberts & Doonan 2016; Roberts 2017).

The latest assessment estimated population sustainability thresholds (PSTs) for New Zealand sea lions at the Auckland Islands, using a population model fitted to demographic observations of females. The

PST criterion was defined by the Ministry for Primary Industries (MPI) as the maximum number of annual mortalities relating to this fishery which would result in a less than 10% probability of a 5% depletion in the mature female population projected over 5 years, relative to that occurring in the absence of fishery-related mortality. This resulted in a PST of 23 females (46 combined sexes) (Roberts 2017). This PST value was then adjusted to account for potential mortalities in other trawl fisheries at the Auckland Islands including for scampi (SCI 6A) to obtain an annual mortality limit of 38 sea lions (combined sex) in the squid trawl fishery (MPI 2017).

The research described in this report updates the demographic population model used to estimate the PST for NZ sea lions in the Auckland Islands squid trawl fishery (SQU 6T), and to estimate the population effects of fishery deaths at different levels. The specific project objectives were as follows:

1. **Update the demographic population model of female New Zealand sea lions at the Auckland Islands**, incorporating: new demographic observations up to the 2019 field season and estimates of historical annual squid trawl fishery deaths derived from a spatial fisheries risk model (PRO2017-10) and assessment of cryptic mortality rates (PMM2018-09). Use these demographic models to estimate demographic rates driving periods of increasing, declining and stable pup production trajectory.

2. Estimate the Auckland Islands female New Zealand sea lion population size in 2019 and use this to **calculate the female New Zealand sea lion PST for the Auckland Islands population**, consistent with the population recovery goal(s) specified by Fisheries New Zealand.

3. **Undertake simulations using demographic models of female New Zealand sea lions at the Auckland Islands to estimate the effects of future squid fishery deaths on sea lion population growth**, assuming alternative scenarios of: historical fishery deaths; future population trajectory (i.e., increasing, decreasing or stable); and future deaths (e.g., consistent with the PST, or a continuation of estimated deaths for the recent period).

2.   METHODS

2.1 Demographic population model

Bayesian demographic population models were developed for female New Zealand sea lions at the Auckland Islands, using the SeaBird demographic modelling software (e.g., Francis & Sagar 2012; Roberts & Doonan 2016). In short, the model simulates the numbers of female sea lions of different ages through time, and fits these numbers to the available demographic information. Population model structures were based on the base case models developed by Roberts (2017) that were modified to facilitate the simulation of future population trajectories consistent with historical observed periods of population increase, decline and relative stability. Models integrated information from mark-resighting, age distribution, and annual pup production estimates, and incorporated: new field-based observations up to 2019; and revised estimates of historical squid trawl fishery-related annual deaths, derived from the outputs of Large et al. (2019) and Meyer (2019).

2.1.1   Observations

Population models were fitted to three sources of demographic information through time: mark-resighting observations; estimates of annual pup production; and the age composition of lactating females.

Annual pup production has been estimated for all Auckland Islands breeding rookeries since 1995, using a consistent methodology (DOC 2019). These estimates were then multiplied by 0.5 to approximate the number of female pups born each year (Table A1-1). A coefficient of variation (CV)

of 0.03 was assumed for all years with a normal distribution, consistent with previous modelling by Roberts (2017).

Childerhouse et al. (2010) used tooth-based ageing to estimate the age distribution of lactating females at Sandy Bay and Dundas at the Auckland Islands in all years from 1998–2001. Following the approach of Roberts & Doonan (2016), a combined-rookery time series of age distribution estimates was obtained by multiplying the estimated proportion-at-age for each of the Sandy Bay and Dundas with the estimated pup production for each respective rookery by year (Table A1-1), summing these (to estimate the total number of breeders by age, i.e. across both rookeries) then recalculating the combined proportions-at-age for the years 1998–2001 (Table A1-2). A multinomial distribution was assumed when fitting to age composition estimates with effective sample size (N) of 200 for 1998, and $N = 1000$ for age compositions in 1999, 2000 and 2001 (consistent with Roberts & Doonan 2016 and Roberts 2017).

The processing of mark resighting observations exactly followed the approach used by Roberts & Doonan (2016) and Roberts (2017). Mark resighting data were extracted from the demographic database maintained by Dragonfly Data Science (Dragonfly 2019). Only females marked as pups were included in the model, which were either branded, 'chipped' (with Passive Integrated Transponder (PIT) tags), or flipper-tagged. A distinction was made between these in the model, because of mark-type differences in annual resighting probability, and the inability for field scientists to distinguish between un-tagged females and females which were flipper-tagged only as pups (i.e. not also branded or chipped), but for which all tags had subsequently fallen off (tag loss). This was required so that tag loss rate could be estimated, which would otherwise create a negative bias in annual survival estimates.

Also following the approach of Roberts & Doonan (2016), 'phantom tagged' female pups were appended to the mark-resighting data to account for pups known to have died prior to the date of tagging. These pups were specified as not having been seen after the year of tagging (to replicate the effect of mortality within the first year). The method for estimating the number of phantom tags was as specified by Roberts & Doonan (2016). The total number of pups marked each year by mark type (and phantom tagged pups) are given in Table A1-3. Annual resights by age group are shown along with model fits (Figure A2-3).

2.1.2   Model structure

The set of states that any sea lion can occupy in a particular year is called the partition. The model partitioned the population into ages 1 to 8+ years (for the 8+ partitioning), or 1 to 15+ years (for the 15+ partitioning), with the last age class being a plus group (no maximum age was specified). Each age class was further partitioned into a number of states depending on pupping status and the number of flipper tags. The partition, therefore, accounted for numbers of sea lions by age, pupping status and number of flipper tags within an annual cycle, where movement between partition states was determined by the transition parameters. Sea lions entered the partition as pups and were removed by mortality.

Two partition schemes were used (Figure 2):

- The **8+ Partitioning** considered that sea lions between ages 0 to 7 are "immature" if they had never pupped (individuals were assumed not to pup until age 4); a sea lion between age 4 and 8+ became a "pupper" if she produced a pup in that year. A sea lion which did not produce a pup before age 8 was considered as "immature", or a "non-pupper" if age 8+. With this partition scheme, the model was able to estimate the relative probability of pupping at age.

- The **15+ Partitioning** was as the 8+ Partitioning (above), except that the plus group was at age 15+. This partitioning allowed for the estimation of annual survival for the senescent age group (15+) ($Surv_{15}$) separately from younger ages.

Accordingly, each re-sighting observation in the mark-resighting dataset was assigned a state based on age and pupping status, as well as the number of remaining tags. The pupping status was derived from observations in the "behaviour" field of the mark-resighting data (Dragonfly 2019). If during a particular year an individual was observed giving birth or nursing a pup, then a "pupper" status was ascribed in that year. If one of the remaining females was observed three times without a pup, then a "non-pupper" status was ascribed. If an individual was observed, but the pupping status could not be ascribed according to these rules above, then "unknown" pupper status was ascribed to that observation (a composite class of puppers and non-puppers; not shown in the partition diagram below). All individuals aged 3 or less were assumed to be unable to produce a pup.
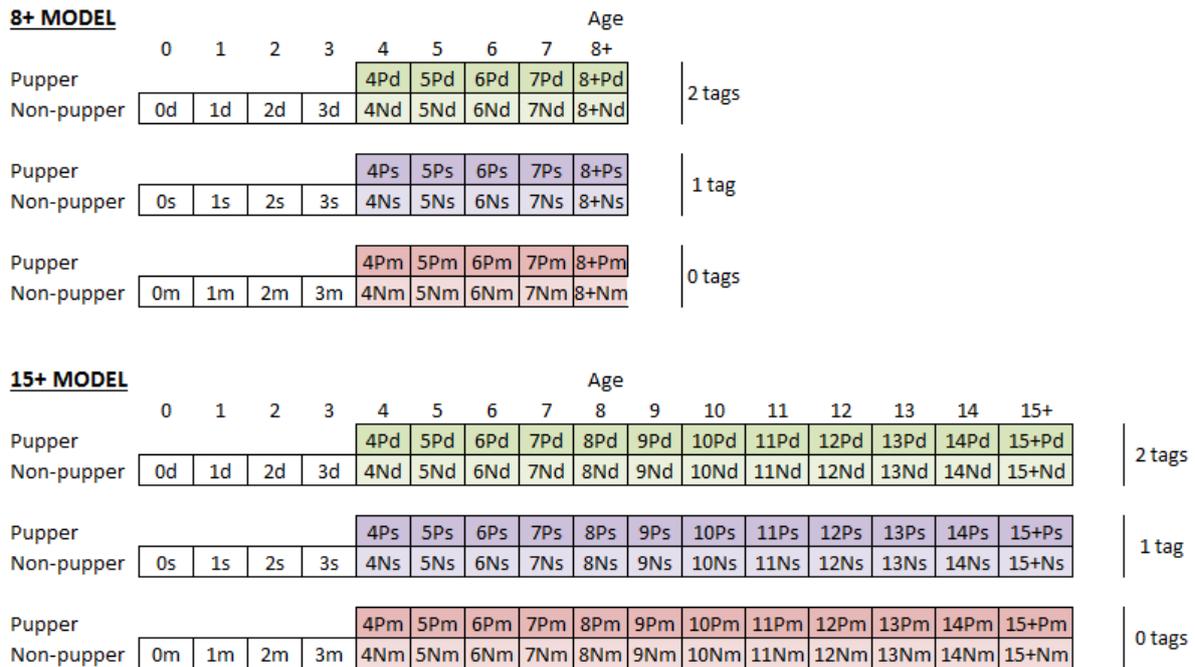
Figure 2: Model partitionings used, including: the '8+ model' (top) used by the base case model configuration 'Yearblock8'; and: the 15+ model (bottom) 'Yearblock15, which has convergence issues at MCMC.

Parameter estimation was by maximum likelihood. Methods used for: parameter estimation, the inclusion of fishery-related deaths, the likelihoods for different observation-types, and the derivation of the objective function also all follow those described in detail by Roberts & Doonan (2016).

2.1.3 Demographic model development and parameterisation

Model parameterisations were modified from the base models used by Roberts & Doonan (2016) and Roberts (2017), with respect to the estimation of annual resighting probability, and the year-blocking of year-varying demographic rates.

Based on MCMC model run estimates by Roberts & Doonan (2016), the annual resighting probabilities of tag and chip-marked females were very similar, i.e., 0.85 (95% CI = 0.77–0.92) for pupping females that were flipper tagged only, and 0.88 (95% CI = 0.80–0.94) for chipped females in the latest model years. As such, the decision was made to use a combined resighting parameter ($Res_{P\_tag\_chip}$) for both of these mark types in order to reduce the potential for confounded model parameters.

The corresponding demographic rates driving periods of different pup production trajectories (i.e., increasing—from 1990 to 1999, decreasing—from 2000 to 2009, or stable—since 2010) were required

for estimating the population effects of future fishery-related death scenarios, given alternative future population trajectories. Thus, key demographic rates were year-blocked, with the breakpoints in survival of immature ages time-lagged to account for the time taken to reach maturity and effect changes in pup production trajectory. The demographic rate year blocks corresponding to each pup production phase (i.e., increasing, decreasing or stable) are given in Table 1. Survival at age 1 ($Surv_1$) was held constant with respect to year, because this would otherwise have been highly correlated with changes in survival in the first year of life ($Surv_0$). Initial model exploration of the data found no evidence for a change in the proportion of females breeding at age 4 (Mat_4) based on model likelihood, so this was also held constant with respect to year.

A list of all parameters estimated by the two models (Yearblock8 and Yearblock15) is displayed in Table 2, along with the year blocks used for year-varying parameters. Uniform priors were assumed for all parameters, bounded at 0 and 1 for all except N_{1990} (the total estimated number of breeders in 1990) (100, 4000), and annual survival parameters, which all used an upper bound of 0.99, consistent with previous modelling by Roberts & Doonan (2016) and Roberts (2017).

Table 1: Year blocks specified for year-varying parameters $Surv_0$ (annual survival to age 1), $Surv_2$ (survival at ages 2-5), $Surv_6$ (survival at ages 6+) and Pr_P (annual pupping rate), consistent with increasing, declining, and stable pup production phases.

| Phases of pup production (years) | Parameters and year blocks used for each pup production phase | | | |
	$Surv_0$	$Surv_2$	$Surv_6$**	Pr_P
Increasing (1990–1999)	1990–1993	1990–2007*	1990–1999	1990–1998
Decreasing (2000–2009)	1994–2005	1990–2007*	2000–2009	1999–2008
Stable (2010–2019)	2006–2018	2008–2018	2010–2018	2009–2018

*The same year period of $Surv_2$ was used for both the increasing and decreasing pup production phases, because the lack of resighting effort prior to 1999 meant that potential changes in this parameter would have been confounded with survival at other ages.
**the same year blocks were used for parameter $Surv_6$, used by the 15+ model.

Table 2: Summary of all estimated model parameters.

Parameter	Description	Year blocks (empty if constant with respect to year)
$Surv_0$	Annual survival age 0	1990–1993, 1994–2005, 2006–2018
$Surv_1$	Annual survival age 1	
$Surv_2$	Annual survival age 2–5	1990–2007, 2008–2018
$Surv_6$*	Annual survival age 6+	1990–1999, 2000–2009, 2010–2018
$Surv_6$**	Annual survival age 6–14	1990–1999, 2000–2009, 2010–2018
$Surv_{15}$**	Annual survival age 15+	
Pr_P	Annual probability of pupping at age 8+	1990–1998, 1999–2008, 2009–2018
Mat_4	Probability of pupping at age 4 (multiplier of Pr_P)	
Mat_5	Probability of pupping at age 5 (multiplier of Pr_P)	
Mat_6	Probability of pupping at age 6 (multiplier of Pr_P)	
Mat_7	Probability of pupping at age 7 (multiplier of Pr_P)	
Res_{1-2}	Annual resighting probability at ages 1–2	
Res_3	Annual resighting probability at age 3	
Res_N	Annual resighting probability of non-puppers	1999, 2000–2001, 2002–2012, 2013, 2014–2015, 2016, 2017, 2018

$Res_{P\_tag\_chip}$	Annual resighting probability of puppers that were not branded as pups	1999, 2000–2001, 2002–2012, 2013, 2014–2015, 2016, 2017, 2018
Res_{P_brand}	Annual resighting probability of puppers branded as pups (forced to equal 1)	
T_{1_0}	Annual probability of losing a single tag in the first year	
T_{1_a}	Functional form parameter that gives the probability of losing 1 tag in a year (1)	
T_{1_b}	Functional form parameter that gives the probability of losing 1 tag in a year (2)	
T_2	Annual probability of losing two tags in a year	
N_{1990}	Total number of breeders in 1990	

*Parameter $Surv_6$ (annual survival at age 6+) was estimated when using the 8+ model partition only.
**Parameters $Surv_6$ (annual survival at age 6–14) and $Surv_{15}$ (annual survival at age 15+) were estimated when using the 15+ model partition only.

2.1.4 Historical fishery deaths

Model runs incorporated estimates of historical annual fishery-related deaths of female New Zealand sea lions in the Auckland Islands southern arrow squid trawl fishery (SQU 6T) for years 1990 to 2019 (fishing seasons 1989/90 to 2018/19). Models incorporating historical trawl mortality produced estimates of annual survival that would have occurred in the absence of that mortality, which can then be used by model projections.

For the years 1993 to 2017, models either used the median or upper 95% CI of annual female capture estimates from a spatially-explicit fisheries risk assessment (SEFRA) model of New Zealand sea lions at the Auckland Islands (Large et al. 2019). The spatial risk model did not produce fishery capture estimates for 2018 and 2019, so, for these years the arithmetic mean of estimates from 2013–2017 was used. For the years 1990 to 1992, the reported squid catch tonnage by fishing season for the SQU 6T fishery (Fisheries New Zealand, 2018, pp. 60) was used to estimate sea lion captures, by calculating the ratio of squid catch in each year relative to 1994 (1993 was not used on the basis of anomalously low squid catch in that year, suggesting that fishing operations were likely to be atypical), and then applying the respective ratio to the 1994 estimate of sea lion captures to estimate sea lion captures in 1990–1992. In order to account for cryptic mortality relating to the use of sea lion exclusion devices (SLEDs), Fisheries New Zealand specified that a cryptic mortality multiplier of 1.5 was then applied to annual captures since 2009 to estimate annual deaths in those years (note that this is slightly higher than the base case cryptic mortality multiplier estimated subsequently by Meyer 2019). This multiplier was informed by the preliminary outputs of separate research estimating mortality relating to the use of SLEDs (Meyer 2019). These annual mortality values were applied to the 3+ age group, with equal vulnerability assumed across these ages (Table 3).

To account for pup mortality relating the mortality of females in the squid trawl fishery, related pup deaths were also incorporated into the model. Annual female pup deaths were calculated by multiplying annual adult female mortality by 0.375 (assuming a 0.75 pupping rate, 50:50 sex ratio and 100% mortality of all pups born to adult females killed) (Table 3).

Table 3: Historical estimated annual deaths at ages 0 (pups) and 3+ used by assessment model runs, based on the median or upper 95% CI of spatial risk model estimates.

	Median female deaths		Upper 95% CI female deaths	
Year	**Age 3+**	**Pups**	**Age 3+**	**Pups**
1990	48.43	18.16	67.47	25.30
1991	25.99	9.75	36.21	13.58
1992	26.48	9.93	36.90	13.84
1993	10.02	3.76	13.53	5.07
1994	84.21	31.58	117.32	44.00
1995	75.73	28.40	105.43	39.54
1996	85.35	32.01	115.58	43.34
1997	52.80	19.80	75.42	28.28
1998	28.85	10.82	38.79	14.55
1999	5.99	2.25	9.21	3.45
2000	19.65	7.37	27.16	10.19
2001	8.96	3.36	12.39	4.65
2002	24.49	9.18	33.44	12.54
2003	23.10	8.66	32.21	12.08
2004	32.23	12.09	45.42	17.03
2005	37.13	13.92	53.26	19.97
2006	32.46	12.17	50.06	18.77
2007	16.66	6.25	23.93	8.97
2008	17.22	6.46	24.60	9.23
2009	6.50	2.44	13.28	4.98
2010	5.04	1.89	7.95	2.98
2011	5.37	2.01	9.98	3.74
2012	2.42	0.91	6.60	2.48
2013	3.42	1.28	6.38	2.39
2014	1.56	0.59	3.12	1.17
2015	1.44	0.54	3.03	1.14
2016	3.14	1.18	6.36	2.39
2017	4.29	1.61	8.07	3.03
2018	2.77	1.04	5.39	2.02
2019	2.77	1.04	5.39	2.02

2.2 Model runs

A list of all model runs undertaken is shown in Table 4. MCMC runs were undertaken for all of these models.

Table 4: A list of all female New Zealand sea lion population model runs undertaken by this assessment.

Model name	Model description
Yearblock8_zero	Uses the Yearblock8 model (with a plus group at age 8+) and does not incorporate historical squid fishery-related deaths.
Yearblock8_median	Uses the Yearblock8 model (with a plus group at age 8+) and median historical squid fishery-related deaths (Table 1).
Yearblock8_upper	As the Yearblock8_median run, except that the upper 95% CI of historical squid fishery-related deaths was used (Table 1).
Yearblock15_zero	Uses the Yearblock15 model (with a plus group at age 15+) and does not incorporate historical squid fishery-related deaths.
Yearblock15_median	Uses the Yearblock15 model (with a plus group at age 15+) and median historical squid fishery-related deaths (Table 1).
Yearblock15_upper	As the Yearblock15_median run, except that the upper 95% CI of historical squid fishery-related deaths was used (Table 1).

2.3 MCMC methods

MCMC methods followed the basic approach of Roberts & Doonan (2016). Samples were taken at 100 iteration intervals along each chain. As a preliminary step to improve mixing, the covariance matrix

used to generate the proposal distribution was iteratively recalculated from the first 200+ samples until the degree of MCMC mixing was adequate. This process was repeated five times, with minimal improvement obtained in mixing from the final iteration.

Three MCMC chains were then run using the optimised covariance matrix for each model run. The chains were longer for the Yearblock8 model runs (140 000 iterations = 1400 samples) than for the Yearblock15 model runs (42 000 iterations = 420 samples) due to an approximate 4-fold difference in runtime. The first 50 samples were discarded from each chain to give a total combined sample size of 4050 for the Yearblock8 model and 1100 samples for the Yearblock15 model (across all three chains for each model run).

2.4 Estimation of the population sustainability threshold (PST)

The method for estimating the female PST for New Zealand sea lions at the Auckland Islands was specified by Fisheries New Zealand. This used the equation adopted for SEFRA risk assessment (Sharp 2018),

$$PST = N \; \frac{1}{2} \; r^{\max} \phi,$$

where N is the estimate of total female population size (the median estimate of the Yearblock8_zero model for the year 2019), $r^{\max}$ is the intrinsic population growth rate (a value of 0.12 was used, consistent with the default used for stock assessment in the U.S. (Wade 1998)), and ϕ is a calibration coefficient that can be tuned to achieve user-specified population-based management goals. For ϕ, Fisheries New Zealand specified a base case value of 0.1 and sensitivity values of 0.05 and 0.20. Future fishery mortality at these values of ϕ are consistent with population recovery to 95.0%, 97.5% or 90.0% of the unimpacted population state, respectively, under a default assumption of linear density dependence.

2.5 Population projections

Projection methods also followed the basic approach of Roberts & Doonan (2016) and Roberts (2017). The operating model and the sets of parameter distribution were used to assess the effect of alternative future fishery-related mortalities on projected mature female population size up to eleven years into the future (i.e., up to the year 2030, which accounts for the population effects of 10 future years of commercial fishing operations). Demographic rates used in the projections were sampled from MCMC estimates.

Population projections were run to estimate the population consequences of alternative levels of future deaths relating to the squid trawl fishery (SQU 6T). Future deaths could only be specified as a constant amount with respect to future year (i.e., rather than a proportion), so that proportion killed would be different for increasing versus declining populations. Future catch level scenarios included: zero future fishery-related mortality ("zero"), future deaths equal to the mean of estimates for 2013–2017 ("average"), or equal to the estimated PST with alternative levels of ϕ (0.05, 0.10 or 0.20). Thus, a total of five alternative future catch scenarios were assessed.

In addition to assuming five alternative levels of future fishery-related deaths, population projections also used alternative:

- Population models (Yearblock8 or Yearblock15)
- Historical squid trawl-related mortalities (median or upper 95% CI)
- Future population growth trajectories (assuming demographic rate year blocks consistent with increasing, decreasing, or stable total pup production trends) (Table 1).

Mature female population size was derived from model estimates of population size by age and demographic status. This was calculated as the model estimate of population size at ages 8+ (all assumed to be mature from age 8) plus the estimated population size at ages 4–7 times the relative pupping rate estimate at ages 4–7 (Mat_4, Mat_5, Mat_6 and Mat_7), e.g., mature female N at age 5 (N_{age5}) was calculated as $N_{age5} \times Mat_5$, etc.

The population effects of future fishery deaths were presented as the effect on the projected annual population growth rate of mature females (λ), or the effect on the projected population status of mature females relative to what was estimated in the absence of any future fishery-related mortality (N_{2025} % no squid fishery deaths).

3.   RESULTS

3.1 Model diagnostics

Model fits are shown for an intermediate model run using the Yearblock8_zero model structure and parameterisation, although without historical fishery-related deaths specified. Maximum posterior density (MPD) model fits to annual pup production, lactating female age composition, and mark resighting observations are shown in Appendix 2. These plots indicate a reasonable model fit to the observations, with minimal conflict from the different sources of information.

The MCMC traces and cumulative frequency plots for demographic parameters are shown in Appendix 3. These were produced for both the Yearblock8_zero and Yearblock15_zero model runs (i.e., not incorporating historical fishery-related deaths). These indicate that MCMC mixing was reasonable for the Yearblock8 model, although it was poor for the Yearblock15 model (or the model run time was too slow and the chain too short, given the degree of mixing). For this reason, the Yearblock8 model run was used as the base case model structure, although parameter estimates and projection statistics were also obtained for the Yearblock15 model run.

3.2 Parameter estimates

Parameter estimates are given in Appendix 4 for all MCMC runs assuming either the median or upper 95% CI of historical fishery-related deaths. There was good agreement comparing parameter estimates from the Yearblock8 and Yearblock15 model runs (comparing Table A4-1 and Table A4-2). As expected, including the senescent survival age group ($Surv_{15}$) in the Yearblock15 model produced lower estimates of survival at ages 6–14 ($Surv_6$) relative to $Surv_6$ from the Yearblock8 model (which included survival at older ages).

Demographic rates that will be directly affected by fishery-related deaths were bolded in Table A4-1 and Table A4-2. As expected, annual survival estimates for vulnerable ages (age 3+) increased as historical mortality levels increased (when additional fishery mortality is incorporated, the model must produce higher survival estimates to fit to the same observations). For the recent period of stable pup production trajectory (2010 to 2019), survival estimates at vulnerable ages was relatively insensitive to the incorporation of historical mortality. For example, $Surv_6$ was 0.887 (95% CI = 0.876–0.896) for the model not incorporating historical fishery mortality, compared with $Surv_{6+}$ of 0.887 (95% CI = 0.876–0.898) for the model incorporating the upper 95% CI of historical deaths (Table A4-1). A much greater effect of historical fishery deaths was estimated for earlier increasing (1990 to 1999) and decreasing (2000 to 2009) pup trajectory phases.

As a by-product, this assessment estimated the changes in demographic rates creating changes in pup production trajectory (i.e., from increasing, to decreasing to stable). Model estimates are consistent with the change from increasing to declining pup production trajectory being driven by:

- A large decrease in survival to age 1 ($Surv_0$ was 0.825 (0.715–0.960) for 1990 to 1993, compared with 0.389 (0.347–0.436) for 1994 to 2004; from the model using the median of historical fishery deaths);

- A large decrease in survival at ages 6+ ($Surv_6$ was 0.957 (0.936–0.977) for 1990 to 1999, compared with 0.857 (0.848–0.866) for 2000 to 2009); and

- A decrease in annual pupping rate (Pr_P was 0.784 (0.719–0.855) for 1990 to 1998, compared with 0.713 (0.683–0.742) for 1999 to 2008);

The decreases in all three parameters can be consider as 'significant' on the basis of non-overlapping 95% credible intervals between time periods.

The more recent changes from decreasing to stable pup production trajectory was estimated to be driven by:

- Increased survival at ages 2–5 ($Surv_2$ was 0.912 (0.889–0.934) for 1990 to 2007, compared with 0.943 (0.923–0.960) for 2008 to 2018); and
- Increased survival at ages 6+ ($Surv_6$ was 0.857 (0.848–0.866) for 2000 to 2009, compared with 0.887 (0.877–0.897) for 2010 to 2018); and
- Slightly increased annual pupping rate (Pr_P was 0.713 (0.683–0.742) for 1999 to 2008, compared with 0.748 (0.718–0.777) for 2009 to 2018).

The estimate of first year survival was very similar comparing the decreasing ($Surv_0$ was 0.389 (0.347–0.436) for 1994 to 2004) and stable pup trajectory phases (0.378 (0.337–0.426) for 2005 to 2018). Also, provisional modelling found no evidence for changes in pupping rate at age 4 (Mat_4) between periods.

3.3 Population size and PST

The total female population size of New Zealand sea lions at the Auckland Islands (i.e., mature and immature) was estimated to be 4,293 individuals in 2019 (95% CI = 4,120–4,473), using the Yearblock8_zero model (without historical fishery mortality incorporated). The equivalent estimate from the Yearblock15_zero model was 4,314 individuals in 2019 (95% CI = 4,129–4,494). Only the Yearblock8_zero estimate was used for the estimation of PST values (Table 5).

Given a current (2019) female population size of 4,293 individuals and the PST equation shown in Section 2.4, female PST values of 13, 26, or 52 females were derived for ϕ values of 0.05, 0.10 and 0.20, respectively.

Table 5: Estimated total female population size by year from the Yearblock8_zero and Yearblock15_zero model runs. Quantiles are the median followed by the 95% credible intervals in parentheses. The 2019 estimate used to estimate the PST is bolded.

Year	Yearblock8	Yearblock15	Year	Yearblock8	Yearblock15
	Quantiles of estimated total female population size			Quantiles of estimated total female population size	
1990	5 174 (4 448–6 051)	6 047 (5 237–7 150)	2020	4 241 (4 058–4 432)	4 260 (4 068–4 452)
1991	5 560 (4 850–6 412)	6 397 (5 607–7 438)	2021	4 189 (3 995–4 393)	4 208 (4 005–4 404)
1992	5 980 (5 284–6 795)	6 761 (6 007–7 749)	2022	4 137 (3 934–4 352)	4 154 (3 942–4 361)
1993	6 425 (5 753–7 204)	7 147 (6 436–8 071)	2023	4 086 (3 873–4 313)	4 098 (3 879–4 319)
1994	6 908 (6 261–7 645)	7 549 (6 883–8 421)	2024	4 036 (3 813–4 276)	4 044 (3 820–4 276)
1995	6 920 (6 373–7 552)	7 440 (6 860–8 174)	2025	3 987 (3 751–4 238)	3 993 (3 764–4 235)
1996	7 027 (6 559–7 560)	7 417 (6 943–8 046)	2026	3 938 (3 691–4 201)	3 941 (3 707–4 194)
1997	7 183 (6 793–7 622)	7 450 (7 080–7 959)	2027	3 889 (3 632–4 163)	3 890 (3 650–4 148)
1998	7 363 (7 034–7 723)	7 492 (7 192–7 920)	2028	3 841 (3 575–4 128)	3 841 (3 591–4 109)
1999	7 544 (7 247–7 848)	7 529 (7 273–7 904)	2029	3 793 (3 518–4 094)	3 792 (3 531–4 070)
2000	7 591 (7 269–7 929)	7 512 (7 243–7 876)	2030	3 747 (3 460–4 059)	3 745 (3 477–4 036)
2001	7 218 (6 925–7 515)	7 194 (6 947–7 523)	2031	3 700 (3 404–4 025)	3 698 (3 416–4 002)
2002	6 863 (6 598–7 124)	6 872 (6 649–7 166)	2032	3 655 (3 351–3 991)	3 652 (3 358–3 967)
2003	6 536 (6 294–6 767)	6 557 (6 358–6 822)	2033	3 611 (3 297–3 956)	3 605 (3 300–3 931)
2004	6 233 (6 009–6 445)	6 255 (6 071–6 480)	2034	3 567 (3 243–3 922)	3 559 (3 246–3 898)
2005	5 949 (5 740–6 146)	5 963 (5 792–6 172)	2035	3 523 (3 189–3 890)	3 514 (3 195–3 863)
2006	5 662 (5 481–5 833)	5 684 (5 523–5 860)	2036	3 480 (3 138–3 859)	3 470 (3 144–3 828)
2007	5 390 (5 224–5 555)	5 414 (5 252–5 573)	2037	3 437 (3 086–3 830)	3 425 (3 092–3 792)
2008	5 129 (4 966–5 295)	5 148 (4 978–5 309)	2038	3 396 (3 036–3 799)	3 382 (3 041–3 755)
2009	4 931 (4 769–5 101)	4 956 (4 786–5 120)	2039	3 354 (2 986–3 767)	2 992 (3 340–3 721)
2010	4 786 (4 617–4 961)	4 841 (4 672–5 016)			
2011	4 733 (4 575–4 898)	4 771 (4 609–4 931)			
2012	4 681 (4 530–4 834)	4 707 (4 552–4 853)			
2013	4 626 (4 479–4 774)	4 647 (4 494–4 790)			
2014	4 569 (4 424–4 717)	4 588 (4 436–4 729)			
2015	4 512 (4 363–4 661)	4 530 (4 377–4 676)			
2016	4 456 (4 304–4 610)	4 476 (4 310–4 623)			
2017	4 401 (4 242–4 561)	4 420 (4 251–4 577)			
2018	4 346 (4 181–4 517)	4 365 (4 191–4 535)			
2019	**4 293 (4 120–4 473)**	4 314 (4 129–4 494)			

3.4 Population projections

Female population projections were undertaken for the scenarios described in Section 2.5, and applying future squid fishery-related death scenarios shown in Table 6.

Table 6: Future constant annual fishery deaths at ages 0 (pups) and 3+ used for model projections.

Scenario	Median historical deaths		Upper 95% CI historical deaths	
	Pups	Age 3+	Pups	Age 3+
Zero	0	0	0	0
Average	1.04	2.77	2.02	5.39
Equal to PST ($\phi = 0.05$)	5	13	5	13
Equal to PST ($\phi = 0.10$)	10	26	10	26
Equal to PST ($\phi = 0.20$)	20	52	20	52

As expected, increasing future fishery-related deaths resulted in progressively lower future population size (from left to right in Figure 3), relative to what was estimated in the absence of future squid fishery-related mortality (comparing black and grey lines in Figure 3). The population effects of alternative

future squid fishery mortality levels were larger in the declining future population growth scenario (top row compared with middle and lower rows of Figure 3).

The effect of alternative future squid fishery deaths scenarios on the annual population growth rate of mature females in 2030 (λ_{2030}) was estimated (Table A5-1). Assuming stable future population growth, median historical fishery squid fishery deaths and future squid fishery deaths equal to the base case PST ($\phi = 0.10$), the estimated value of λ_{2030} was 0.978 (0.973–0.983), compared with 0.989 (0.985–0.993) in the absence of future squid fishery deaths. The scenario using the upper 95% CI of historical deaths, estimated very similar values of λ_{2030} (comparing the top and bottom half of Table A5-1).

Yearblock15 model estimates of λ_{2030} were virtually the same as those of the Yearblock8 model for the declining and stable population growth scenarios (comparing Table A5-1 and Table A5-2). The Yearblock15 model estimated a lower growth rate for the increasing period, relative to the Yearblock8 model, although the population effects of alternative future deaths were comparable across the two models.

Plots of projected mature female population status given alternative future fishery death scenarios (relative to that estimated in the absence of future fishery deaths) are shown in Figure 4. Table A5-3 summarises the mature female population status in 2025 for all projection scenarios. For the model run assuming median historical fishery squid fishery deaths, stable future population growth, and future squid fishery deaths equal to the base case PST ($\phi = 0.10$), the estimated mature female population size in 2025 was 95.4% (95% CI = 95.1%–95.6%) of that estimated in the absence of future fishery deaths. Comparable estimates were obtained when assuming population decrease or increase in the future (median population status ranged from 95.0% to 96.1% of unimpacted levels). A sensitivity run with future deaths equal to the PST with $\phi = 0.05$ resulted in a mature female population size in 2025 of 97.7% of the unimpacted level (95% CI = 97.6%–97.8%), i.e., halving future mortalities produced a proportional improvement in future population status.

When future squid fishery deaths were equal to the average across the most recent estimated period, the mature female population status in 2025 was 99.5% of that estimated in the absence of future squid fishery mortality (95% CI = 99.5%–99.5%). This status was relatively insensitive to the future population growth scenario (Table A5-3).
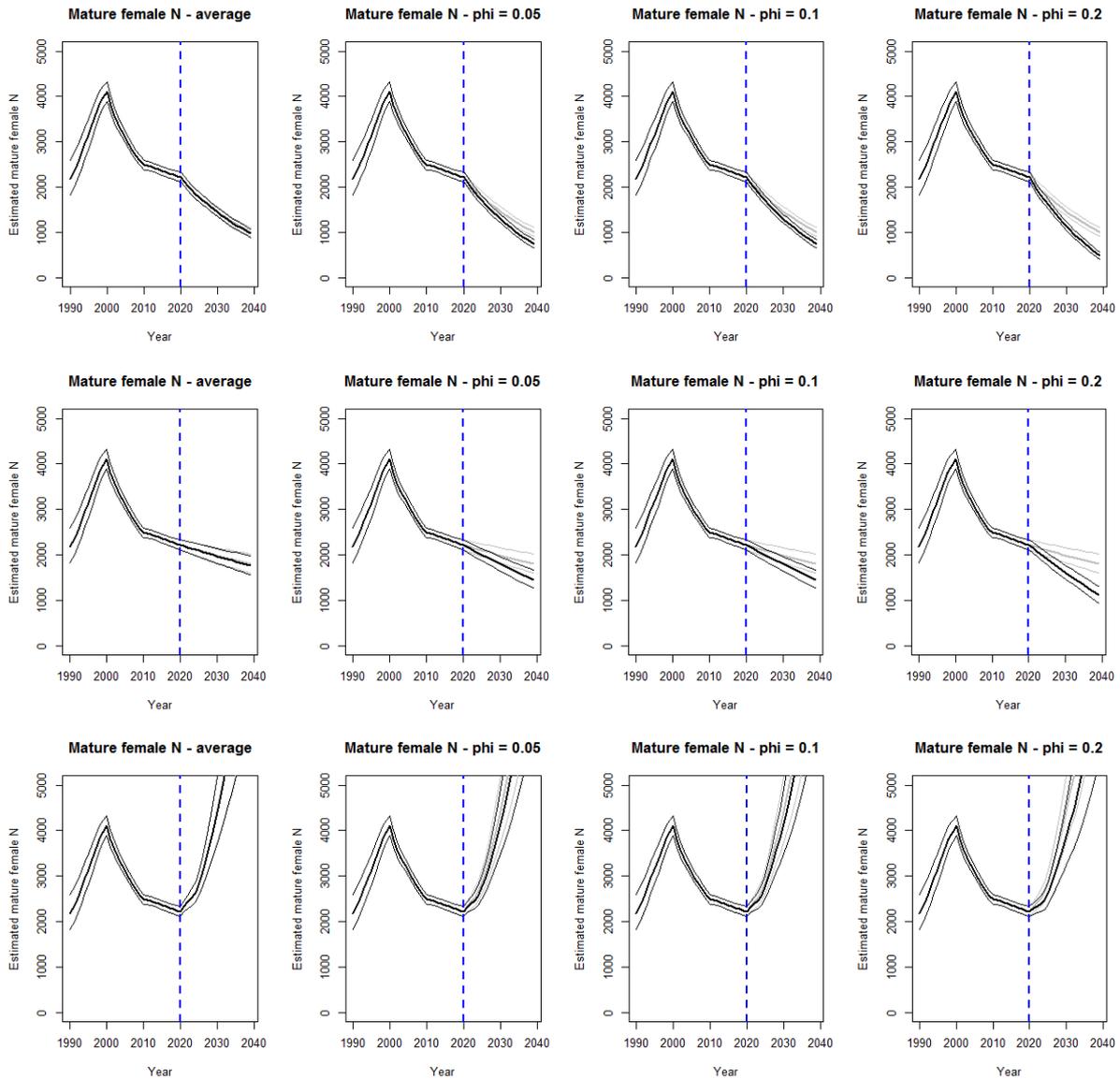
Figure 3: Female New Zealand sea lions population projections with alternative scenarios of: population growth (i.e., decreasing (top row), stable (middle row) or increasing (bottom row)); and future squid fishery-related deaths (i.e., zero future deaths (grey, shown in all plots for reference), the average of the last five years' estimated deaths, or equal to the population sustainability threshold (PST) assuming alternative values of ϕ ("phi") (all black)). Simulations were obtained using the 'Yearblock8' population model.
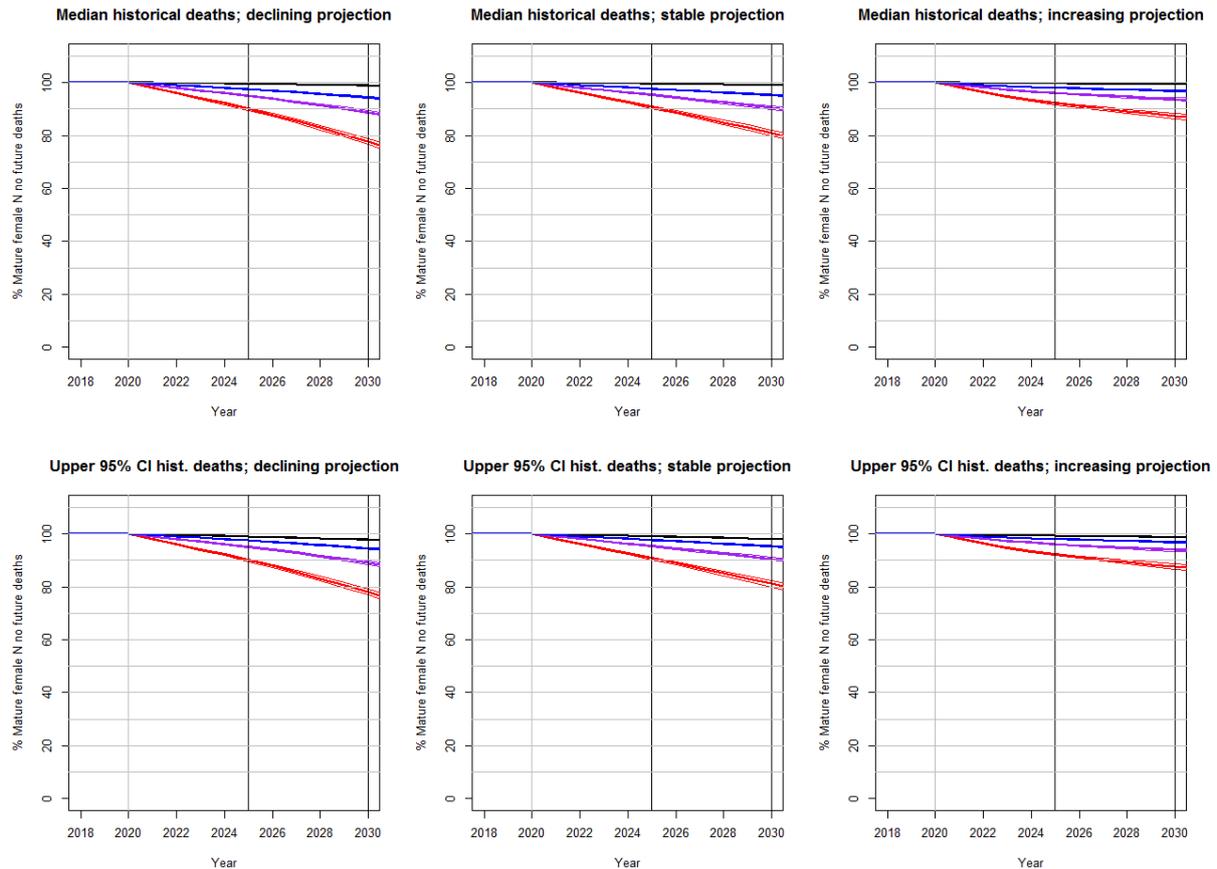
Figure 4: Projected population size of female New Zealand sea lions at the Auckland Islands expressed as a percentage of projected population size under the zero future squid fishery (SQU 6T) related deaths scenario (N_{2025} % no squid fishery deaths), with alternative scenarios of: population growth (i.e., decreasing (left), stable (middle) or increasing (right)); historical deaths relating to the (median (top row) or upper 95% of spatial risk model estimates (bottom row)); and future deaths relating to this fishery (i.e., the average of the last five years' estimated deaths (black), or equal to the population sustainability threshold (PST) assuming $\phi = 0.05$ (blue), $\phi = 0.10$ (purple), or $\phi = 0.20$ (red). Simulations were obtained using the 'Yearblock8' population model.

4. DISCUSSION

4.1 Limitations of data and modelling approach

The assessment models were fitted to demographic data collected at the Auckland Islands since 1990, including annual pup production estimates (all Auckland Islands rookeries), age composition of reproductive females and mark-resighting observations (both Sandy Bay only). Apart from annual pup production, all demographic information came from the Sandy Bay population. However, this rookery only supports 19% of the total Auckland Islands pup production, with the much larger Dundas Island population contributing 77% of the total (based on 2019 estimates) (DOC 2019). A large number of female pups have now been tagged at Dundas Island since the early 1980s, although resighting effort is typically deemed too low to obtain robust estimates of key demographic rates. A single year of concerted resighting effort at Dundas Island would provide key information about relative cohort strength, which could be used to assess the representativeness of Sandy Bay demographics. Note that an initial exploration by Roberts et al. (2014) indicated some similarities in first year survival comparing the Sandy Bay and Dundas Island rookeries. Multiple years of resighting effort at Dundas would provide the data requirements for including Dundas Island mark resighting observations in future population assessments.

All population models were developed using NIWA's SeaBird demographic modelling program (e.g. Francis & Sagar 2012). This program offers considerable flexibility with respect to specifying alternative parameterisations and the specification of different life history processes, although it cannot currently be configured to represent density dependent changes in demographic rates. SeaBird can, however, simulate the population effects of assuming alternative future catch scenarios (point values of number of deaths by year), which will be complementary to other assessments including density dependence. Note that the method used here to estimate the PST assumes density dependent population growth rate, which changes with population status relative to carrying capacity (K) (Sharp 2018).

The large number of demographic states required to represent tag loss, maturation (and senescence for the Yearblock15 model) (Figure 2) meant that model transition matrices were large. This meant that MCMC chains were slow to run. In addition, SeaBird uses the Metropolis–Hastings MCMC algorithm, which can result in poor mixing when there are correlated parameters (as there often are in demographic assessment models). A workaround was found to deal with this issue (iterative recalculation of the covariance matrix from initial MCMC samples), although future assessments should potentially consider alternative MCMC algorithms that improve MCMC mixing (e.g., Hamiltonian Monte Carlo).

The Yearblock8 model used half as many states as the Yearblock15 model (Figure 2) but produced very similar estimates of population status (comparing Table A5-3 and Table A5-4), and MCMC mixing was much improved (see Appendix 3). Future assessment models may not need to use the Yearblock15 model structure.

4.2 Population Sustainability Threshold

Fisheries New Zealand specified a revised criterion for estimating the PST for New Zealand sea lions at the Auckland Islands. Previously this was based on the mortality level which would not reduce the mature female population status by more than 5% over 5 years. A PST of 23 females (46 combined sex) was estimated using this criterion (Roberts 2017). The updated PST criterion used the SEFRA approach with a base case ϕ of 0.10, which would result in recovery to 95% of population size at equilibrium (if future deaths were equal to the PST). The female PST value using this method was 26 females (52 combined sex, assuming equal sex ratio)—close to the previous estimate.

A point estimate of the PST was required for estimating future catch limits, rather than a distribution, as is typically used for estimating risk using the SEFRA method (e.g. Abraham et al. 2017; Roberts et al. 2019). In addition to ϕ, the PST calculation required input values of r^{max} and total female population size (N). The value of N used to derive the PST was the median total female population size estimate in 2019 (4293 individuals). The credible interval around this estimate was narrow (4129–4494 individuals) such that using the lower 95% CI value would have resulted in a PST of 24 females, close to the base case PST value of 26. The high precision of this estimate will mainly be driven by mark resighting observations confirming individual pupping status. Mark resighting observations inform the estimation of annual pupping rate, for obtaining the breeding population size, given annual pup production. This highlights the need for consistent mark resighting field methods through time with good numbers of breeding status confirmations each year.

In lieu of any robust estimates of r^{max} for New Zealand sea lions, this assessment used the default value of 0.12 used for pinnipeds in U.S. stock assessments (Wade 1998). Rapid population growth has been observed in New Zealand sea lion populations at Campbell Island, the NZ mainland and potentially also Stewart Island, which could be used to estimate minimum values of r^{max}. However, life history invariant approaches (e.g., Dillingham et al. 2016) may be preferable for estimating r^{max} given the rich demographic information available for adopting this approach, including from both growing and declining populations (e.g., Roberts & Doonan 2016).

4.3 Population effects of fisheries deaths

For the period of standardised SLED use (since 2009), annual deaths of female New Zealand sea lions in the Auckland Islands were estimated by applying a cryptic morality multiplier of 1.5 to the spatial risk model estimate of annual captures (as directed by Fisheries New Zealand). The cryptic mortality multiplier of 1.5 was informed by preliminary outputs of Bayesian simulations by separate research (Meyer 2019). This resulted in an average of 2.77 female deaths annually from 2013 to 2017, or 5.39 females using the upper 95% CI of spatial risk model captures. These estimates are, respectively, around a tenth and a fifth of the estimated annual PST of 26 females (using the base case ϕ of 0.10). Future deaths at these levels were, respectively, estimated to result in a mature female population size in 2025 of 99.5% or 99.0% of what was estimated in the absence of future fishing mortality, with narrow credible intervals (see Table A5-3).

Subsequently to this analysis, a SEFRA model accounting for cryptic mortality relating to SLED-use estimated 2.33 annual deaths in the squid trawl fishery (upper 95% CI = 5.00) from 2014/15 to 2016/17 (Large et al. 2019). These values are marginally lower than those used in the population models developed for this assessment and, so, using them instead would have had a very small effect on model outputs.

This research only assessed the population effects of estimated deaths in the southern arrow squid trawl fishery, i.e., it did not consider other commercial trawl targets around the Auckland Islands, including: scampi, hoki, and other targets. Of these, only the Auckland Islands scampi fishery had comparable estimates of annual New Zealand sea lion deaths to the squid fishery, and is likely to have exceeded squid fishery captures of females in recent years (Large et al. 2019). The annual PST estimated by this research (26 females, using the base case ϕ of 0.10) could ultimately be applied to all Auckland Islands fisheries (and potentially include other anthropogenic causes of death).

The spatial risk assessment by Large et al. (2019) also found evidence for a strong female bias in captures by the scampi fishery. New Zealand sea lions are polygamous, and population productivity will be much more responsive to changes in female population size than of males. Accordingly, nearly all of the foraging information collected from Auckland Islands sea lions to date was from females (Large et al. 2019), and the most recent demographic models and risk assessments for this population are female only (Large et al. 2019; Roberts 2017). This supports the current approach of estimating a female PST in the first instance, and then modifying this estimate to include males as well (e.g., doubling the female PST, under the assumption of an equal sex ratio).

4.4 Demographic drivers of population change

As a by-product, the assessment model estimated the demographic causes of observed changes in pup production trajectory. With respect to the change from increasing to decreasing trajectory after the late 1990s (see Table A1-1), this assessment re-iterated the findings of Roberts & Doonan (2016), who identified multiple demographic drivers including large reductions in pup and adult survival as well as annual pupping rate.

The current assessment was the first to assess the demographic causes of the change from declining to stable trajectory and found that this was primarily driven by increased survival of both sub-adults (age 2–5) and adults (age 6+). Estimated pup survival (to age 1) was estimated to have remained below 40% since 2005 ($Surv_0 = 0.378$ (95% CI = 0.337–0.426)), far below the estimate for years 1990–1993 ($Surv_0$ = 0.825 (95% CI = 0.715–0.960)).

Demographic scenarios by Roberts & Doonan (2016) indicate that pup survival rate can be highly influential for the mature female population trajectory, and this may need to improve in order to achieve population growth. The demographic scenarios by Roberts & Doonan (2016) also indicated that an increasing trajectory may also be achieved by improvements to adult female survival. However, the population assessment reported here strongly suggests that the required increase in adult survival could

not be achieved through the alleviation of squid fishery mortality alone (see the left-hand column of Figure 3).

5. CONCLUSIONS

The primary conclusions of this research were as follows:

- A Bayesian demographic population model was developed for female New Zealand sea lions at the Auckland Islands, integrating demographic information from estimates of age composition, annual pup production, and mark resighting data up to the 2019 field season.

- Population models incorporated estimates of historical annual deaths of New Zealand sea lions in the Auckland Islands squid trawl fishery (SQU 6T), which were derived by applying a cryptic mortality rate of 1.5 times annual captures (based on the provisional outputs of separate research).

- The revised population model of female New Zealand sea lions at the Auckland Islands estimated a current population size of 4293 females in 2019 (95% CI = 4120–4473).

- This population estimate was used to derive an annual female PST of 26 females (using the base case ϕ of 0.10). Assuming this value of ϕ, future fishery mortality equal to the PST would be consistent with population recovery to at least 95% of the unimpacted population state, under a default assumption of linear density dependence.

- Future squid trawl fishery mortalities equal to the base case PST (ϕ of 0.10) were estimated to result in a mature female population size in 2025 of between 95.0% (95% CI = 94.7–95.2%) and 96.1% (95% CI = 95.8–96.3%) of what would have occurred in the absence of incidental mortality from the squid fishery (depending on the future population growth scenario).

  This compares with future squid trawl fishery deaths equal to recent estimated levels which were estimated to result in a mature female population size in 2025 of 99.5% of that estimated in the absence of future squid fishery mortality (95% CI = 99.5%–99.5%). Specifying future deaths at the upper 95% of recent estimate squid fishery deaths resulted in a slightly less optimistic status in 2025 of 99.0% of unimpacted levels (95% CI = 99.0%–99.1%).

- In lieu of a male, or combined sex population model, female PSTs may be doubled to estimate a combined sex annual PST of 52 individuals.

- The primary demographic driver of the recent change in pup production trajectory after 2009, from decreasing to stable, appears to be increased survival at ages 2 and older, but the pupping rate may have increased slightly also. Estimates of pup survival (up to age 1) were very similar across these two periods.

6. ACKNOWLEDGMENTS

7. REFERENCES

Abraham, E.R.; Neubauer, P.; Berkenbusch, K.; Richard, Y. (2017). Assessment of the risk to New Zealand marine mammals from commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report 189*. 123 p.

Augé, A. (2010). Foraging ecology of recolonising female New Zealand sea lions around the Otago Peninsula, New Zealand. Ph.D. Thesis, University of Otago.

Baker, C.S.; Boren, L.; Childerhouse, S.; Constantine, R.; van Helden, A.; Lundquist, D.; Rayment, W.; Rolfe, J.R. (2019). Conservation status of New Zealand marine mammals, 2019. *New Zealand Threat Classification Series 29*. 18 p.

Breen, P.A.; Fu, D.; Gilbert, D.J. (2016). Sea lion population modelling and management procedure evaluations. *New Zealand Aquatic Environment and Biodiversity Report No. 175*. 89 p.

Childerhouse S.J.; Dawson S.M.; Slooten E.; Fletcher D.J.; Wilkinson I.S. (2010). Age distribution of lactating New Zealand sea lions: Interannual and intersite variation. *Marine Mammal Science, 26*: 123–139.

Chilvers, B.L. (2019). Report of the survey to assess pup production of New Zealand sea lions on Stewart Island 2019. 3 p. URL https://www.doc.govt.nz/globalassets/documents/conservation/native-animals/marine-mammals/nz-sea-lion-tmp/louise-chilvers-stewart-island-report-2018-2019.pdf

Dillingham, P.W.; Moore, J.E.; Fletcher, D.; Cortes, E.; Curtis, K.A.; James, K.C.; Lewison, R.L. (2016). Improved estimation of intrinsic growth r max for long-lived species: integrating matrix model and allometry. *Ecological Applications 26*: 322–333.

[DOC] Department of Conservation (2019). New Zealand sea lion monitoring and pup production at the Auckland Islands 2018/19 research report. Report produced by the Department of Conservation. 32 p. URL https://www.doc.govt.nz/globalassets/documents/conservation/marine-and-coastal/marine-conservation-services/reports/pop2018-03-sea-lion-pup-count-2018-19.pdf

Dragonfly (2019). New Zealand sea lion demographic. Database maintained by Dragonfly Data Science Wellington. Retrieved 5th April 2019 from https://sealions.dragonfly.co.nz/demographics/.

Fisheries New Zealand (2018). Fisheries Assessment Plenary, May 2018: stock assessments and stock status. Compiled by the Fisheries Science and Information Group, Fisheries New Zealand, Wellington, New Zealand. 1674 p.

Fletcher, D. (2004). Review of modelling approaches for the New Zealand sea lion PMP. Proteus Wildlife Research Consultants, 8 November 2004. WGNHO-207998.doc. URL https://deepwatergroup.org/wp-content/uploads/2013/09/Fletcher-et-al.-2004-Review-of-Modelling-Approaches-for-the-New-Zealand-Sea-Lion-PMP.pdf

Fyfe, J. (2019). Otago [New Zealand sea lion] pup births. Presentation to the CSP TMP sea lion Technical Working Group, 26 March 2019. URL https://www.doc.govt.nz/contentassets/6886b020667a45599b32e696ff777203/jim-fyfe-otago-pup-numbers-2018-2019.pdf

Francis, R.I.C.C.; Sagar, P.M. (2012). Modelling the effect of fishing on southern Buller's albatross using a 60-year dataset. *New Zealand Journal of Zoology, 39*: 3–17.

Large, K.; Roberts, J.; Francis, M.; Webber, D.N. (2019). Spatial assessment of fisheries risk for Auckland Islands New Zealand sea lions. *New Zealand Aquatic Environment and Biodiversity Report No. 224*.

Meyer, S. (2019). Desktop estimation of New Zealand sea lion cryptic mortality in trawls using SLEDs. *New Zealand Aquatic Environment and Biodiversity Report No. 222*. 25 p.

Meyer, S.; Robertson, B.C.; Chilvers, B.L.; Krkošek, M. (2015). Population dynamics reveal conservation priorities of the threatened New Zealand sea lion *Phocarctos hookeri*. *Marine Biology, 162*: 1587–1596.

[MPI] Ministry for Primary Industries (2017). Squid 6T Operational Plan. Decision document. Report produced by the Ministry for Primary Industries, Wellington. 32 p. URL https://www.mpi.govt.nz/dmsdocument/26551/send

Roberts, J. (2017). Supplemental sea lion population modelling to support an updated Squid Trawl Fishery Operational Plan – Draft Report. MPI Final Research Report for Project SEA2016-24. 10 p.

Roberts, J.; Doonan, I. (2016). Quantitative Risk Assessment of Threats to New Zealand Sea Lions. *New Zealand Aquatic Environment and Biodiversity Report No. 166*. 111 p.

Roberts, J.; Fu, D.; Doonan, I.; Francis, C. (2014). NZ sea lion: demographic assessment of the causes of decline at the Auckland Islands. Demographic model options: demographic assessment. Report produced by NIWA for the Department of Conservation. 142 p. URL https://www.doc.govt.nz/Documents/conservation/marine-and-coastal/marine-conservation-services/reports/NZ-sea-lion-demographic-assessment-causes-decline-auckland-islands-part-1.pdf

Roberts, J.; O'Driscoll, R.L.; Hart, A.; Graham, B. (2018). Survey of New Zealand sea lion prey at the Auckland Islands and Stewart Snares Shelf. *New Zealand Aquatic Environment and Biodiversity Report No. 204*. 64 p.

Roberts, J.; Webber, D.N.; Roe, W.D.; Edwards, C.T.T.; Doonan, I.J. (2019). Spatial risk assessment of threats to Hector's and Māui dolphins (*Cephalorhynchus hectori*). *New Zealand Aquatic Environment and Biodiversity Report No. 214*. 168 p.

Roe, W.D.; Rogers, L.; Pinpimai, K.; Dittmer, K.; Marshall, J.; Chilvers, B.L. (2015). Septicaemia and meningitis caused by infection of New Zealand sea lion pups with a hypermucoviscous strain of *Klebsiella pneumoniae*. *Veterinary Microbiology, 176*: 301–308.

Sharp, B.R. (2018). Spatially-explicit fisheries risk assessment (SEFRA): a framework for quantifiying and managing incidental commercial fisheries impacts on non-target species. In: Aquatic Environment and Biodiversity Annual Review 2017. Ministry for Primary Industries. pp. 20–56. URL https://www.mpi.govt.nz/dmsdocument/27471-aquatic-environment-and-biodiversity-annual-review-aebar-2017-a-summary-of-environmental-interactions-between-the-seafood-sector-and-the-aquatic-environment.

Stewart-Sinclair, P. (2013). The role of long-term diet change in the decline of the New Zealand sea lion population. M.Sc. Thesis, Massey University, New Zealand.

Wade, P. (1998). Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Marine Mammal Science, 14*: 1–37.

Weir, J.; Foo, D.; Buckle, K.; Campbell, J.; Alegre, M.; Scott, N.; (2019). New Zealand Sea Lion TMP: Campbell Island 2018 – 2019. Presentation to the CSP TMP sea lion Technical Working Group, 26 March 2019. URL https://www.doc.govt.nz/contentassets/6886b020667a45599b32e696ff777203/jody-weir-nz-sea-lion-campbell-island-fieldwork-2018-2019.pdf

APPENDIX 1 – OBSERVATIONS FITTED TO BY POPULATION MODELS

Table A1-1: Annual female New Zealand sea lions pup production estimates for the Auckland Islands (1995 to 2019), fitted to by population models in the assessment.

Year	Female pup production estimate	Year	Female pup production estimate
1995	1 259.0	2008	1 087.5
1996	1 342.5	2009	750.5
1997	1 487.5	2010	907.0
1998	1 510.5	2011	775.0
1999	1 433.5	2012	842.0
2000	1 428.0	2013	934.0
2001	1 429.5	2014	787.5
2002	1 141.0	2015	788.0
2003	1 258.5	2016	863.5
2004	1 257.5	2017	982.5
2005	1 074.0	2018	896.0
2006	1 044.5	2019	839.0
2007	1 112.0		

Table A1-2: Estimated age composition of female, lactating New Zealand sea lions at the Auckland Islands (1998 to 2001), fitted to by population models in the assessment. These estimates were combined from Sandy Bay and Dundas Island observations, rescaled for relative pup production (see Roberts & Doonan 2016 for more details).

			Year					Year	
Age	1998	1999	2000	2001	Age	1998	1999	2000	2001
3	0	0	0	0	15	0.03	0.051	0.051	0.045
4	0.03	0.02	0.029	0.03	16	0.059	0.017	0.038	0.051
5	0.059	0.037	0.032	0.043	17	0.011	0.031	0.054	0.026
6	0.1	0.094	0.052	0.034	18	0	0.019	0.02	0.041
7	0.167	0.071	0.071	0.068	19	0	0.026	0.033	0.033
8	0.059	0.127	0.097	0.078	21	0	0.019	0.014	0.016
9	0.089	0.107	0.124	0.124	22	0	0.008	0.014	0.019
10	0.16	0.076	0.062	0.09	23	0	0.006	0.011	0.015
11	0.041	0.11	0.087	0.086	24	0	0	0.025	0.018
12	0.082	0.031	0.061	0.077	25	0	0.018	0.001	0
13	0.052	0.077	0.077	0.063	26	0	0.006	0.006	0.001
14	0.059	0.048	0.03	0.041					
					Total sample size	43	286	287	292

Table A1-3: Number of female New Zealand sea lions marked at Sandy Bay, Auckland Islands in each study year from 1990–2018, grouped by mark type.

Year	Number of physically marked females			Number of 'phantom' marked individuals			Total number of marked individuals, fitted to by models		
	Flipper-tagged only	Chipped but not branded	Branded	Flipper-tagged only	Chipped but not branded	Branded	Flipper-tagged only	Chipped but not branded	Branded
1990	148	0	0	11	0	0	159	0	0
1991	191	0	0	14	0	0	205	0	0
1992	226	0	0	16	0	0	242	0	0
1993	194	0	0	14	0	0	208	0	0
1994	0	0	0	0	0	0	0	0	0
1995	0	0	0	0	0	0	0	0	0
1996	0	0	0	0	0	0	0	0	0
1997	0	0	0	0	0	0	0	0	0
1998	0	255	0	0	5	0	0	260	0
1999	0	211	0	0	16	0	0	227	0
2000	0	103	137	0	6	5	0	109	142
2001	0	268	0	0	17	0	0	285	0
2002	0	159	0	0	33	0	0	192	0
2003	204	0	0	33	0	0	237	0	0
2004	233	0	0	16	0	0	249	0	0
2005	222	0	0	15	0	0	237	0	0
2006	209	0	0	19	0	0	228	0	0
2007	196	0	0	10	0	0	206	0	0
2008	-11	0	0	11	0	0	0*	0	0
2009	150	0	0	6	0	0	156	0	0
2010	0	170	0	0	9	0	0	179	0
2011	0	179	0	0	9	0	0	188	0
2012	0	181	0	0	9	0	0	190	0
2013	0	168	0	0	8	0	0	176	0
2014	76	81	0	1	2	0	77	83	0
2015	0	144	0	0	4	0	0	148	0
2016	0	167	0	0	7	0	0	174	0
2017	0	155	0	0	9	0	0	164	0
2018	0	156	0	0	11	0	0	167	0

*Pups marked in 2008 were not used by the assessment model due to suspected anomalously high tag loss rate of this cohort, following the approach of Roberts & Doonan (2016).

APPENDIX 2 – MAXIMUM POSTERIOR DENSITY RUN DIAGNOSTICS
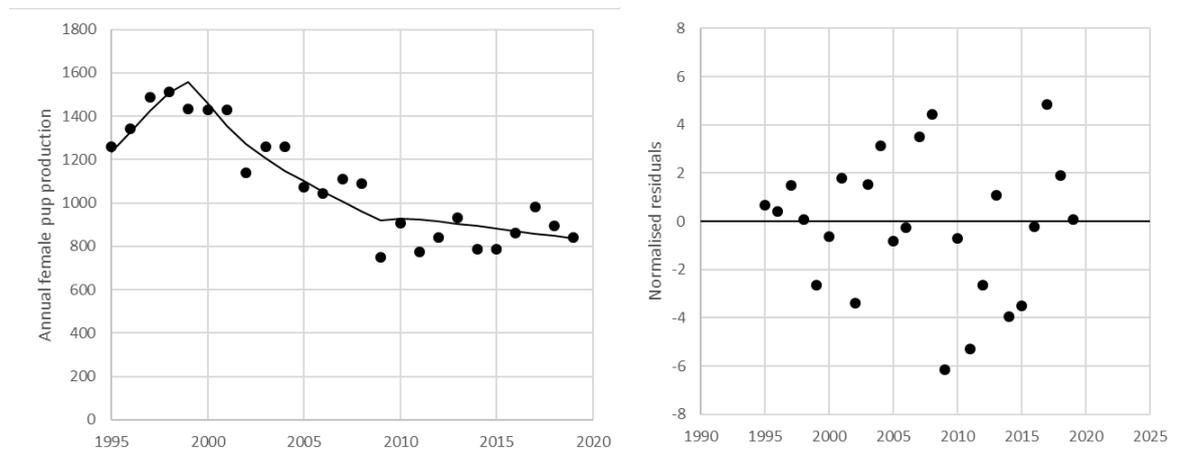
Figure A2-1: Fit of Yearblock8_zero model to annual pup production estimates of female New Zealand sea lions at the Auckland Islands (left); and normalised residuals for the same model fit (right).
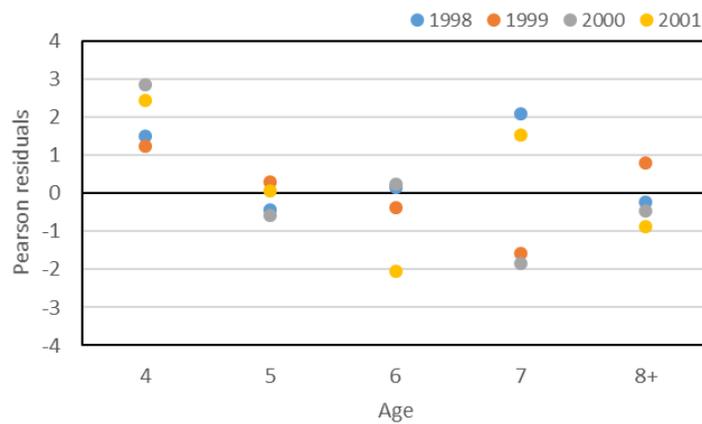

Figure A2-2: Pearson residuals of Yearblock8_zero model fit to the estimated annual age composition of lactating female New Zealand sea lions at Auckland Islands.
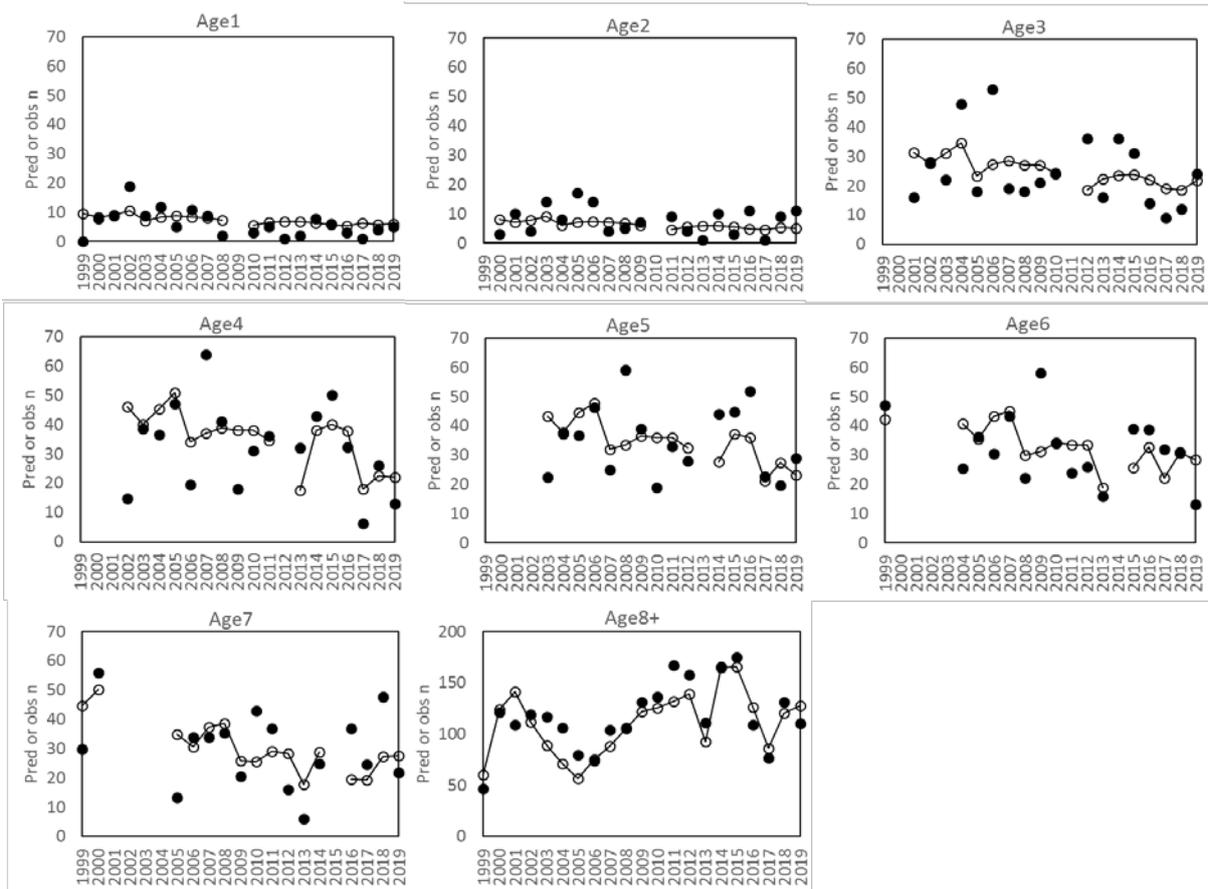
Figure A2-3: Fit of Yearblock8_zero model to annual resighted female New Zealand sea lions at the Auckland Islands, by year and age group.

APPENDIX 3 – MCMC RUN DIAGNOSTICS

MCMC traces for all estimated parameters (<u>8+ model</u>, not incorporating historical fishery-related deaths)
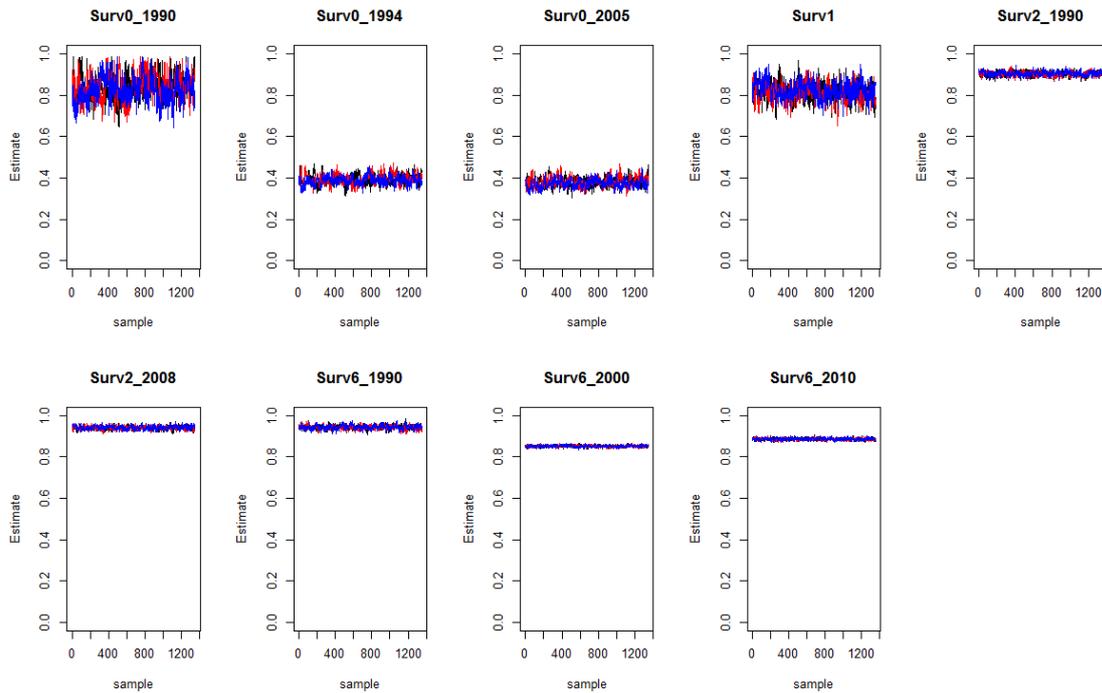
Figure A3-1: MCMC traces for all estimated survival parameters from the <u>Yearblock8_zero</u> model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter.
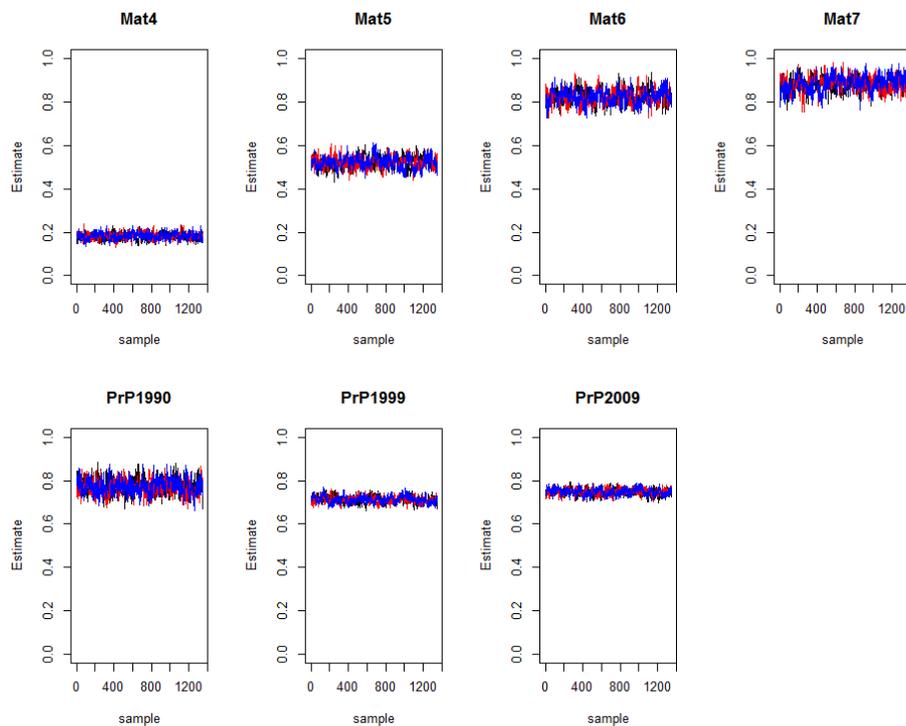
Figure A3-2: MCMC traces for all estimated pupping parameters from the <u>Yearblock8_zero</u> model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter.

Figure A3-3: MCMC traces for all estimated annual resighting parameters from the Yearblock8_zero model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter.
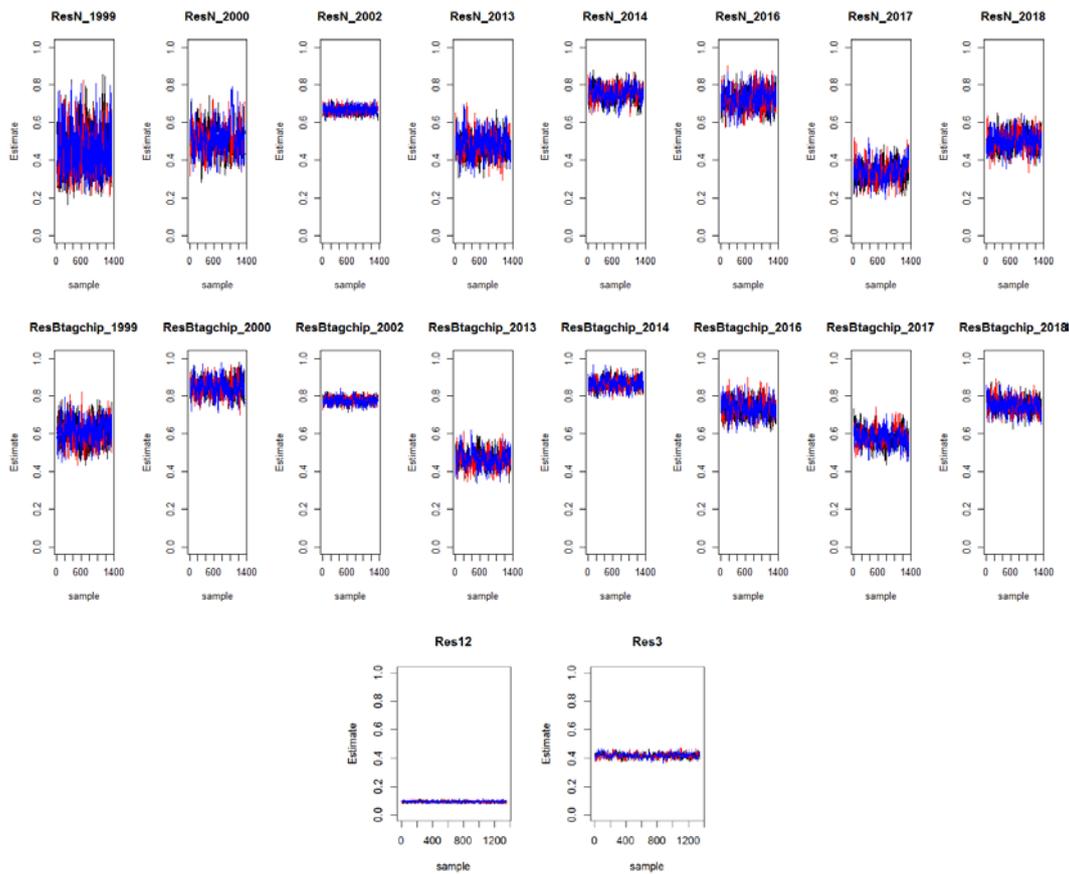

Figure A3-4: MCMC traces for all estimated tag loss rate parameters from the Yearblock8_zero model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter.

Figure A3-5: MCMC traces for the population size scalar parameter from the Yearblock8_zero model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter.
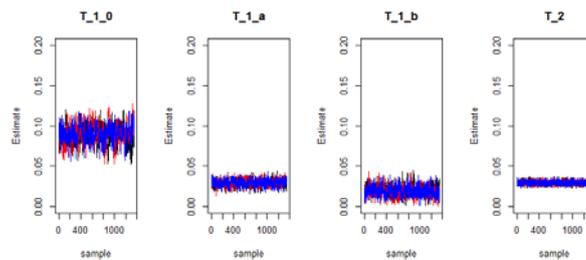
Cumulative frequency plots of MCMC traces for estimated survival and breeding parameters (8+ model, not incorporating historical fishery-related deaths)

Figure A3-6: Cumulative frequency of MCMC traces for all estimated survival parameters from the Yearblock8_zero model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter.
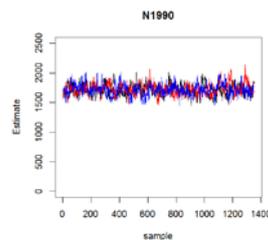

Figure A3-7: Cumulative frequency of MCMC traces for all estimated pupping parameters from the Yearblock8_zero model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter.
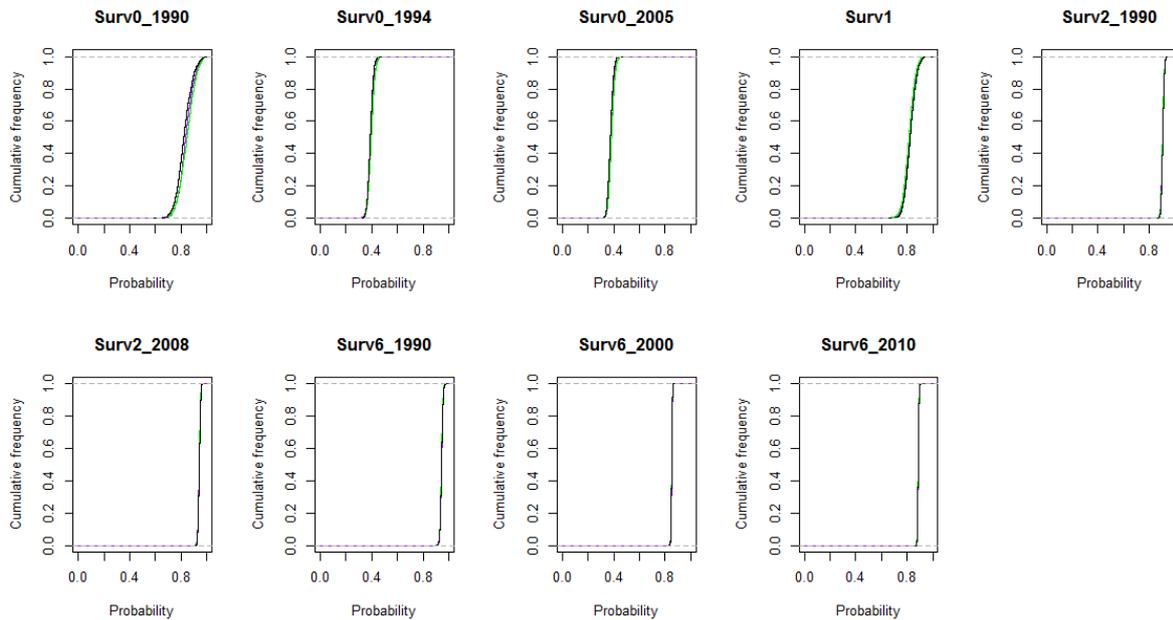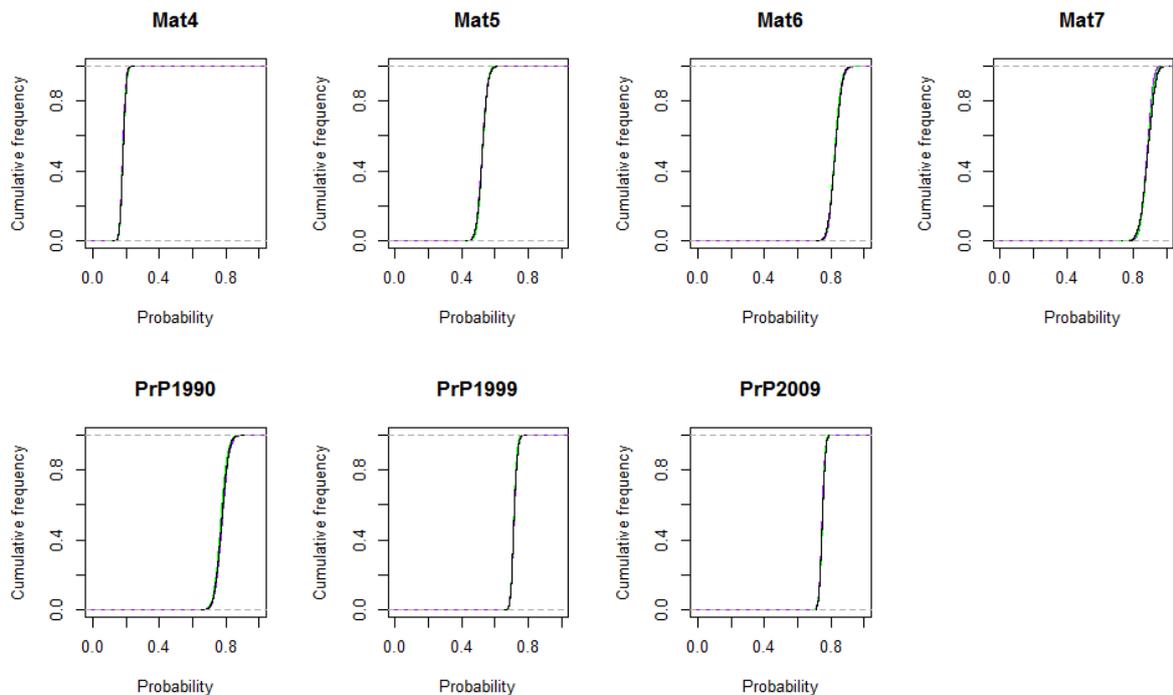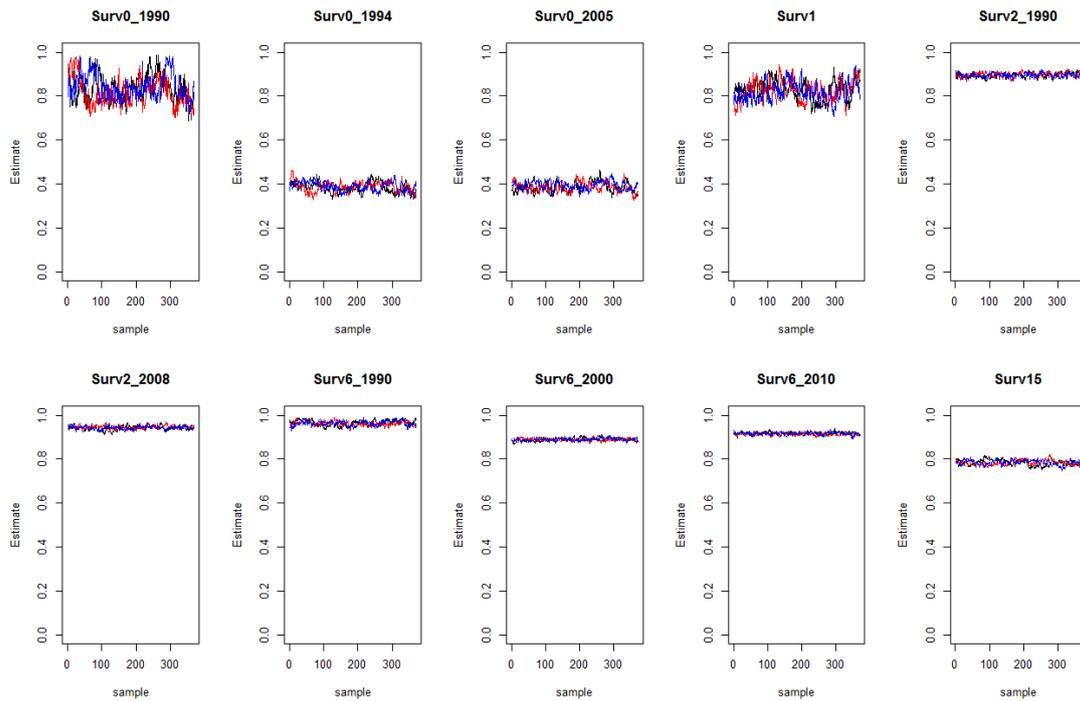
MCMC traces for all estimated parameters (15+ model, not incorporating historical fishery-related deaths)

Figure A3-8: MCMC traces for all estimated survival parameters from the Yearblock15_zero model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter. Note that this model was not adopted as the base case due to poor mixing.

Figure A3-9: MCMC traces for all estimated pupping parameters from the Yearblock15_zero model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter. Note that this model was not adopted as the base case due to poor mixing.

Figure A3-10: MCMC traces for all estimated annual resighting parameters from the <u>Yearblock15_zero</u> model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of th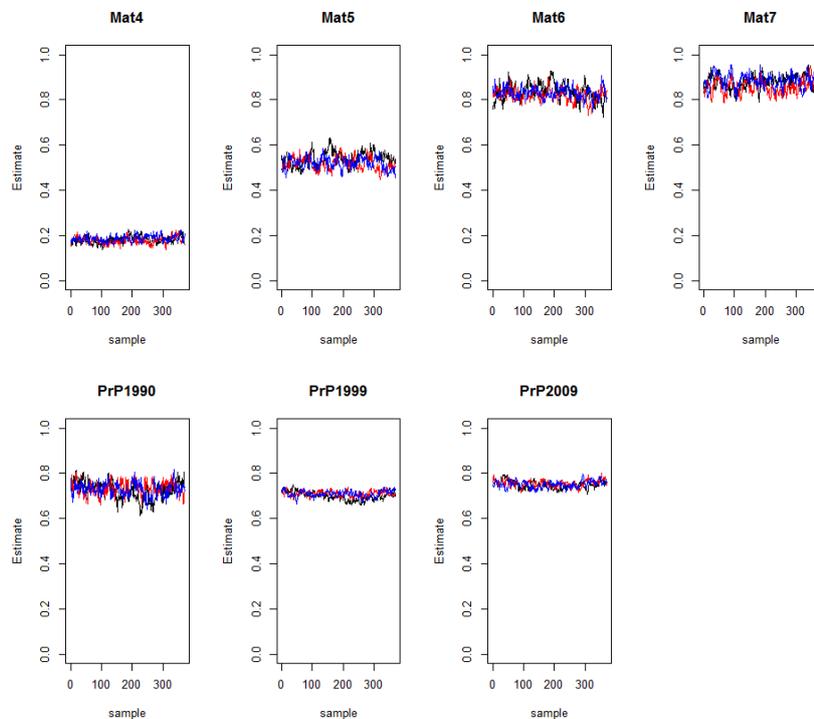ree chains. See Table 2 for a description of each parameter. Note that this model was not adopted as the base case due to poor mixing.

Figure A3-11: MCMC traces for all estimated tag loss rate parameters from the <u>Yearblock15_zero</u> model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter. Note that this model was not adopted as the base case due to poor mixing.
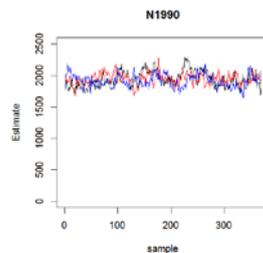
Figure A3-12: MCMC traces for the population size scalar parameter from the <u>Yearblock15_zero</u> model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 1 for a description of each parameter. Note poor mixing for this model run.

Cumulative frequency plots of MCMC traces for estimated survival and breeding parameters (<u>15+ model</u>, not incorporating historical fishery-related deaths)
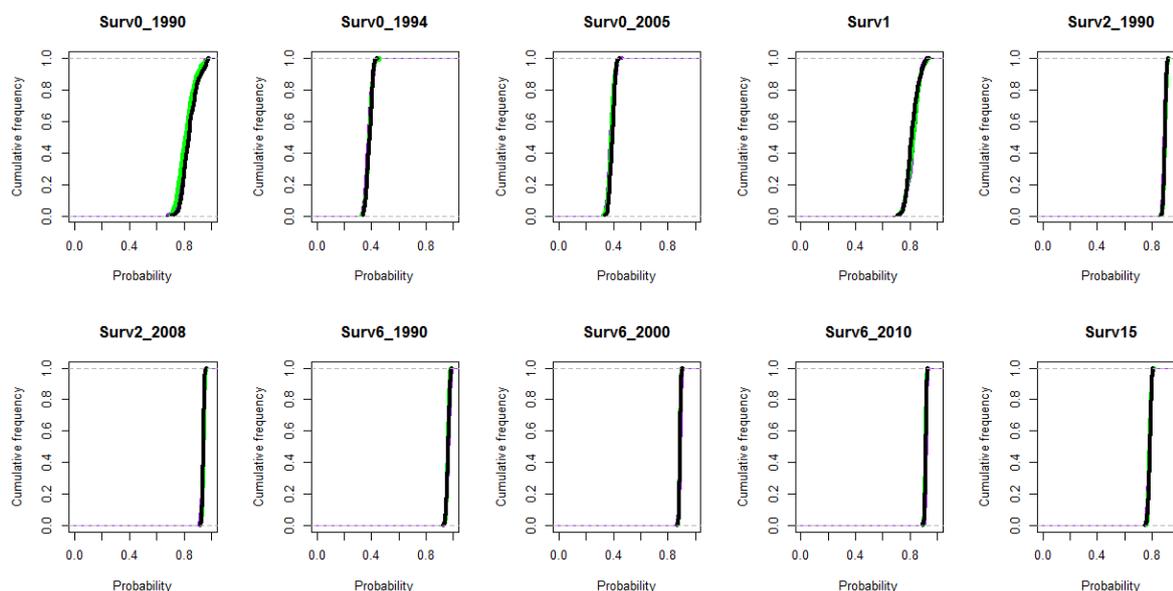
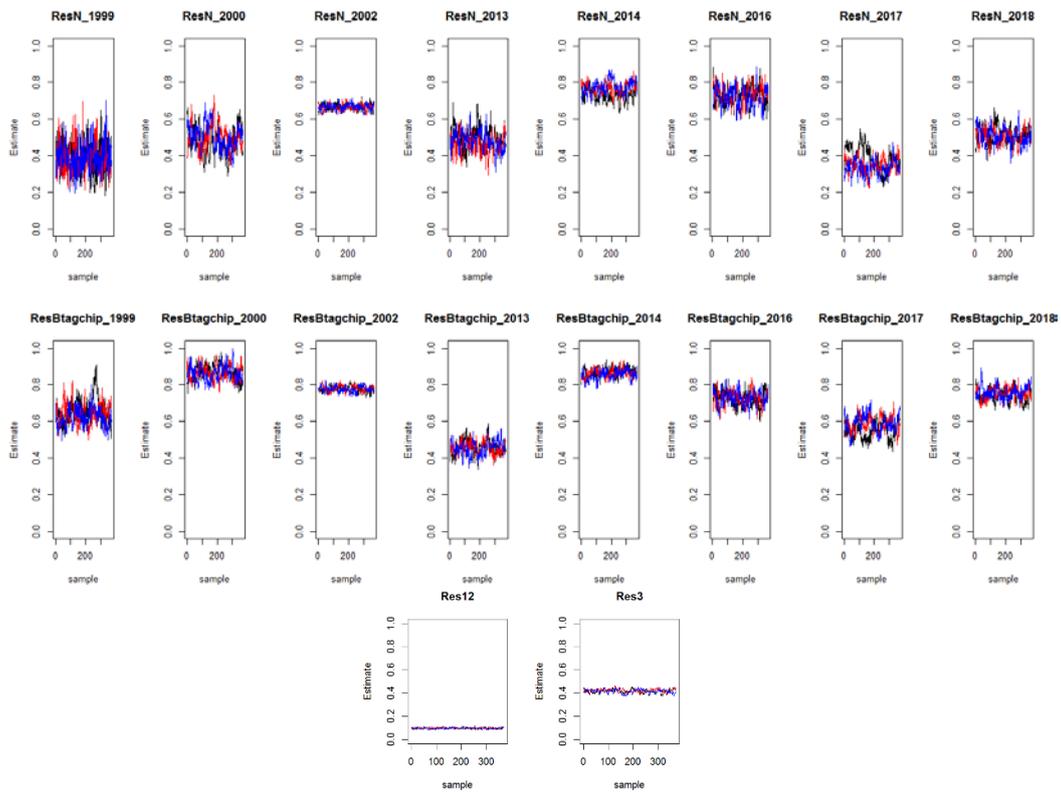**Figure A3-13: Cumulative frequency of MCMC traces for all estimated survival parameters from the <u>Yearblock15_zero</u> model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter. Note poor mixing for this model run.**

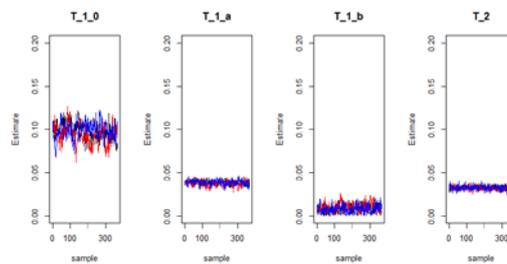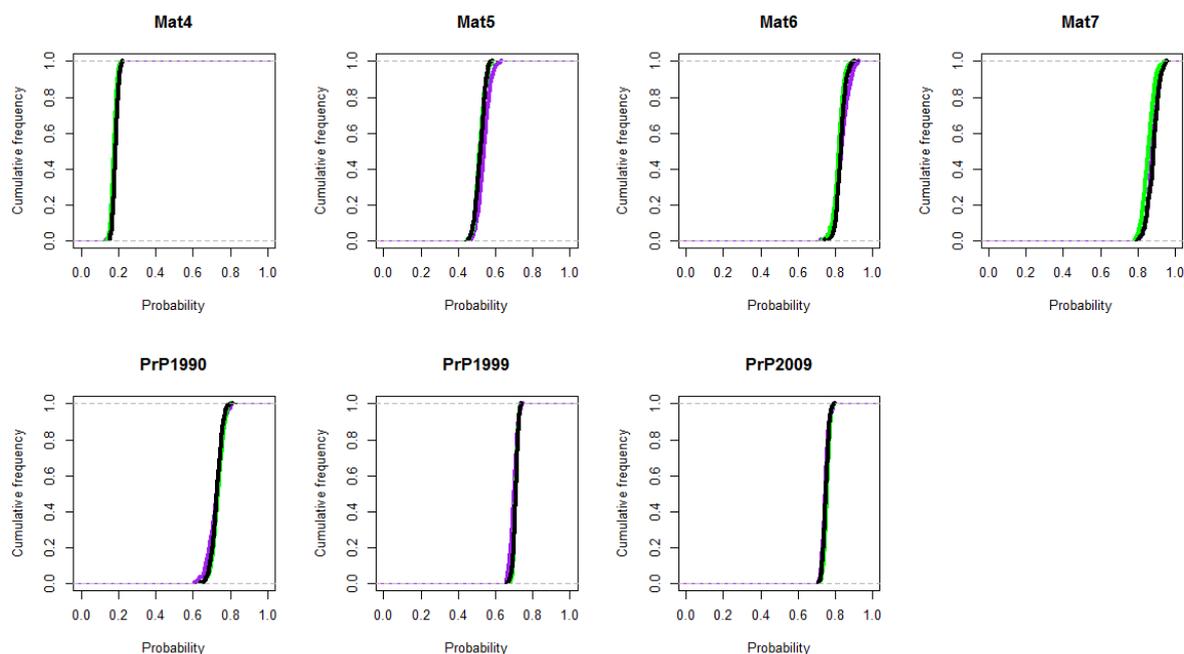**Figure A3-14: Cumulative frequency of MCMC traces for all estimated pupping parameters from the <u>Yearblock15_zero</u> model of female New Zealand sea lions at the Auckland Islands. Different colours were used for each of three chains. See Table 2 for a description of each parameter. Note poor mixing for this model run.**

APPENDIX 4 – MCMC PARAMETER ESTIMATES

Table A4-1: Summary of all estimated parameter values (median and 95% CI) obtained from an MCMC run of the "<u>Yearblock8</u>" population model of female New Zealand sea lions at the Auckland Islands, under alternative assumptions of historical deaths relating to the southern arrow squid trawl fishery at the Auckland Islands (SQU 6T) (zero, median or upper 95% of spatial risk model estimates). Demographic rates that will be directly affected by fishery captures are bolded.

| Parameter | Historical fishery-related deaths scenario | | |
	Zero	**Median**	**Upper**
N_{1990}	1 712 (1 538–1 913)	1 708 (1 539–1 900)	1 712 (1 542–1 900)
$Surv_{0\_1990–1993}$	0.834 (0.719–0.958)	0.825 (0.715–0.960)	0.836 (0.719–0.970)
$Surv_{0\_1994–2004}$	0.392 (0.349–0.441)	0.389 (0.347–0.436)	0.395 (0.353–0.442)
$Surv_{0\_2005–2018}$	0.379 (0.339–0.425)	0.378 (0.337–0.426)	0.383 (0.340–0.434)
$Surv_1$	0.818 (0.735–0.903)	0.827 (0.746–0.916)	0.817 (0.729–0.908)
$Surv_{2\_1990–2007}$	**0.905 (0.884–0.926)**	**0.912 (0.889–0.934)**	**0.916 (0.895–0.936)**
$Surv_{2\_2008–2018}$	**0.941 (0.924–0.958)**	**0.943 (0.923–0.960)**	**0.945 (0.928–0.963)**
$Surv_{6\_1990–1999}$	**0.943 (0.923–0.963)**	**0.957 (0.936–0.977)**	**0.961 (0.941–0.982)**
$Surv_{6\_2000–2009}$	**0.852 (0.842–0.861)**	**0.857 (0.848–0.866)**	**0.859 (0.849–0.869)**
$Surv_{6\_2010–2018}$	**0.887 (0.876–0.896)**	**0.887 (0.877–0.897)**	**0.887 (0.876–0.898)**
$Surv_{15+}$	–	–	–
Mat_4	0.181 (0.153–0.212)	0.185 (0.154–0.216)	0.185 (0.153–0.218)
Mat_5	0.523 (0.471–0.575)	0.525 (0.468–0.578)	0.526 (0.476–0.582)
Mat_6	0.824 (0.764–0.889)	0.825 (0.764–0.885)	0.817 (0.752–0.880)
Mat_7	0.883 (0.813–0.946)	0.888 (0.826–0.945)	0.879 (0.818–0.945)
$Pr_{P\_1990–1998}$	0.773 (0.707–0.841)	0.784 (0.719–0.855)	0.791 (0.724–0.862)
$Pr_{P\_1999–2008}$	0.712 (0.684–0.745)	0.713 (0.683–0.742)	0.714 (0.682–0.742)
$Pr_{P\_2009–2018}$	0.748 (0.717–0.775)	0.748 (0.718–0.777)	0.748 (0.715–0.776)
$Res_{1–2}$	0.096 (0.084–0.108)	0.097 (0.085–0.110)	0.097 (0.085–0.110)
Res_3	0.419 (0.387–0.451)	0.419 (0.388–0.450)	0.419 (0.389–0.453)
Res_{N_1999}	0.444 (0.271–0.687)	0.472 (0.291–0.765)	0.469 (0.291–0.743)
$Res_{N\_2000–2001}$	0.508 (0.382–0.666)	0.511 (0.390–0.663)	0.518 (0.385–0.675)
$Res_{N\_2002–2012}$	0.668 (0.633–0.704)	0.667 (0.631–0.703)	0.667 (0.632–0.702)
Res_{N_2013}	0.479 (0.372–0.594)	0.478 (0.369–0.597)	0.479 (0.379–0.588)
$Res_{N\_2014–2015}$	0.757 (0.673–0.827)	0.759 (0.679–0.835)	0.762 (0.684–0.837)
Res_{N_2016}	0.726 (0.622–0.823)	0.725 (0.621–0.820)	0.722 (0.615–0.822)
Res_{N_2017}	0.335 (0.244–0.442)	0.339 (0.251–0.450)	0.328 (0.229–0.434)
Res_{N_2018}	0.503 (0.415–0.594)	0.503 (0.416–0.606)	0.508 (0.419–0.602)
$Res_{P\_tag\_chip\_1999}$	0.612 (0.501–0.723)	0.606 (0.500–0.721)	0.607 (0.504–0.709)
$Res_{P\_tag\_chip\_2000–2001}$	0.847 (0.766–0.931)	0.840 (0.755–0.926)	0.843 (0.759–0.919)
$Res_{P\_tag\_chip\_2002–2012}$	0.777 (0.744–0.809)	0.777 (0.745–0.810)	0.776 (0.746–0.810)
$Res_{P\_tag\_chip\_2013}$	0.464 (0.385–0.550)	0.468 (0.389–0.556)	0.463 (0.389–0.559)
$Res_{P\_tag\_chip\_2014–2015}$	0.865 (0.814–0.920)	0.865 (0.812–0.921)	0.863 (0.811–0.919)
$Res_{P\_tag\_chip\_2016}$	0.739 (0.655–0.823)	0.739 (0.653–0.816)	0.742 (0.661–0.823)
$Res_{P\_tag\_chip\_2017}$	0.583 (0.503–0.670)	0.751 (0.679–0.821)	0.591 (0.506–0.687)
$Res_{P\_tag\_chip\_2018}$	0.749 (0.680–0.823)	0.091 (0.069–0.114)	0.750 (0.674–0.832)
T_{1_0}	0.089 (0.066–0.112)	0.029 (0.020–0.038)	0.090 (0.066–0.112)
T_{1_a}	0.029 (0.021–0.038)	0.019 (0.005–0.033)	0.029 (0.019–0.038)
T_{1_b}	0.019 (0.006–0.033)	0.029 (0.024–0.034)	0.019 (0.006–0.034)
T_2	0.029 (0.025–0.034)	0.029 (0.025–0.034)	0.029 (0.025–0.034)

Table A4-2: Summary of all estimated parameter values (median and 95% CI) obtained from an MCMC run of the "Yearblock15" population model of female New Zealand sea lions at the Auckland Islands, under alternative assumptions of historical deaths relating to the southern arrow squid trawl fishery at the Auckland Islands (SQU 6T) (zero, median or upper 95% of spatial risk model estimates). Demographic rates that will be directly affected by fishery captures are bolded.

Parameter	Historical fishery-related deaths scenario		
	Zero	**Median**	**Upper**
N_{1990}	1 925 (1 744–2 143)	1 974 (1 807–2 192)	1 972 (1 827–2 168)
$Surv_{0\_1990–1993}$	0.826 (0.729–0.965)	0.787 (0.695–0.895)	0.831 (0.726–0.942)
$Surv_{0\_1994–2004}$	0.386 (0.343–0.429)	0.382 (0.337–0.428)	0.386 (0.349–0.435)
$Surv_{0\_2005–2018}$	0.387 (0.348–0.430)	0.381 (0.344–0.428)	0.383 (0.343–0.435)
$Surv_1$	0.824 (0.741–0.913)	0.843 (0.759–0.922)	0.828 (0.731–0.913)
$Surv_{2\_1990–2007}$	**0.896 (0.875–0.916)**	**0.905 (0.885–0.928)**	**0.906 (0.884–0.928)**
$Surv_{2\_2008–2018}$	**0.944 (0.925–0.961)**	**0.945 (0.927–0.959)**	**0.948 (0.931–0.963)**
$Surv_{6\_1990–1999}$	**0.963 (0.942–0.985)**	**0.974 (0.952–0.989)**	**0.976 (0.955–0.989)**
$Surv_{6\_2000–2009}$	**0.889 (0.876–0.903)**	**0.892 (0.879–0.905)**	**0.896 (0.882–0.910)**
$Surv_{6\_2010–2018}$	**0.916 (0.903–0.928)**	**0.916 (0.904–0.930)**	**0.918 (0.905–0.931)**
$Surv_{15+}$	**0.784 (0.762–0.805)**	**0.786 (0.763–0.809)**	**0.792 (0.770–0.815)**
Mat_4	0.181 (0.153–0.214)	0.182 (0.155–0.215)	0.183 (0.154–0.217)
Mat_5	0.526 (0.472–0.587)	0.519 (0.466–0.576)	0.524 (0.472–0.572)
Mat_6	0.828 (0.768–0.895)	0.813 (0.749–0.875)	0.825 (0.755–0.887)
Mat_7	0.877 (0.809–0.937)	0.861 (0.802–0.932)	0.869 (0.812–0.931)
$Pr_{P\_1990–1998}$	0.731 (0.663–0.787)	0.739 (0.685–0.803)	0.731 (0.683–0.786)
$Pr_{P\_1999–2008}$	0.707 (0.670–0.734)	0.715 (0.685–0.744)	0.705 (0.677–0.733)
$Pr_{P\_2009–2018}$	0.751 (0.722–0.783)	0.756 (0.728–0.784)	0.752 (0.724–0.779)
$Res_{1–2}$	0.096 (0.084–0.108)	0.097 (0.087–0.110)	0.097 (0.085–0.109)
Res_3	0.418 (0.384–0.446)	0.417 (0.387–0.448)	0.422 (0.392–0.455)
Res_{N_1999}	0.391 (0.239–0.567)	0.431 (0.275–0.652)	0.403 (0.257–0.601)
$Res_{N\_2000–2001}$	0.497 (0.356–0.635)	0.513 (0.395–0.691)	0.492 (0.358–0.645)
$Res_{N\_2002–2012}$	0.668 (0.634–0.700)	0.666 (0.633–0.701)	0.668 (0.634–0.704)
Res_{N_2013}	0.482 (0.377–0.593)	0.482 (0.373–0.590)	0.483 (0.379–0.588)
$Res_{N\_2014–2015}$	0.759 (0.678–0.836)	0.755 (0.677–0.826)	0.766 (0.684–0.829)
Res_{N_2016}	0.730 (0.624–0.832)	0.745 (0.651–0.843)	0.729 (0.624–0.837)
Res_{N_2017}	0.350 (0.260–0.476)	0.343 (0.260–0.455)	0.340 (0.237–0.467)
Res_{N_2018}	0.506 (0.419–0.598)	0.497 (0.421–0.583)	0.505 (0.420–0.601)
$Res_{P\_tag\_chip\_1999}$	0.641 (0.543–0.787)	0.644 (0.539–0.748)	0.653 (0.534–0.765)
$Res_{P\_tag\_chip\_2000–2001}$	0.866 (0.790–0.950)	0.841 (0.755–0.923)	0.859 (0.775–0.951)
$Res_{P\_tag\_chip\_2002–2012}$	0.779 (0.744–0.810)	0.777 (0.744–0.811)	0.776 (0.745–0.812)
$Res_{P\_tag\_chip\_2013}$	0.458 (0.381–0.535)	0.453 (0.384–0.534)	0.467 (0.388–0.557)
$Res_{P\_tag\_chip\_2014–2015}$	0.863 (0.810–0.910)	0.861 (0.794–0.909)	0.861 (0.805–0.913)
$Res_{P\_tag\_chip\_2016}$	0.731 (0.644–0.807)	0.721 (0.640–0.806)	0.723 (0.654–0.803)
$Res_{P\_tag\_chip\_2017}$	0.576 (0.478–0.672)	0.569 (0.498–0.662)	0.579 (0.473–0.668)
$Res_{P\_tag\_chip\_2018}$	0.751 (0.683–0.822)	0.754 (0.686–0.836)	0.752 (0.673–0.818)
T_{1_0}	0.097 (0.074–0.117)	0.095 (0.072–0.115)	0.098 (0.077–0.120)
T_{1_a}	0.039 (0.032–0.044)	0.038 (0.031–0.044)	0.039 (0.032–0.044)
T_{1_b}	0.008 (0.001–0.019)	0.009 (0.001–0.023)	0.008 (0.001–0.021)
T_2	0.032 (0.028–0.037)	0.032 (0.028–0.037)	0.032 (0.028–0.036)

APPENDIX 5 – POPULATION PROJECTION OUTPUTS

Table A5-1: Projected annual population growth rate of female New Zealand sea lions at the Auckland Islands in 2030 (λ_{2030}), with alternative scenarios of: population growth (i.e., decreasing, stable or increasing); historical deaths relating to the southern arrow squid trawl fishery at the Auckland Islands (SQU 6T) (median or upper 95% of spatial risk model estimates); and future deaths relating to this fishery (i.e., zero future deaths, the average of the last five years' estimated deaths, or equal to the population sustainability threshold (PST) assuming alternative values of ϕ). Population simulations were obtained using the 'Yearblock8' population model.

Future squid fishery deaths	λ_{2030} Decreasing	Stable	Increasing
Median historical deaths			
zero	0.959 (0.955–0.963)	0.989 (0.985–0.993)	1.087 (1.069–1.105)
average	0.958 (0.954–0.962)	0.988 (0.984–0.992)	1.086 (1.068–1.104)
$\phi = 0.05$	0.953 (0.948–0.957)	0.984 (0.979–0.988)	1.084 (1.066–1.103)
$\phi = 0.10$	0.945 (0.940–0.950)	0.978 (0.973–0.983)	1.082 (1.063–1.101)
$\phi = 0.20$	0.927 (0.921–0.933)	0.965 (0.959–0.971)	1.077 (1.057–1.096)
Upper 95% CI historical deaths			
zero	0.962 (0.957–0.966)	0.990 (0.986–0.995)	1.092 (1.073–1.110)
average	0.959 (0.955–0.964)	0.988 (0.984–0.993)	1.091 (1.072–1.110)
$\phi = 0.05$	0.955 (0.951–0.960)	0.985 (0.980–0.990)	1.089 (1.071–1.109)
$\phi = 0.10$	0.948 (0.943–0.953)	0.980 (0.974–0.985)	1.087 (1.068–1.107)
$\phi = 0.20$	0.931 (0.925–0.937)	0.967 (0.960–0.973)	1.082 (1.062–1.102)

Table A5–2: Projected annual population growth rate of female New Zealand sea lions at the Auckland Islands in 2030 (λ_{2030}), with alternative scenarios of: population growth (i.e., decreasing, stable or increasing); historical deaths relating to the southern arrow squid trawl fishery at the Auckland Islands (SQU 6T) (median or upper 95% of spatial risk model estimates); and future deaths relating to this fishery (i.e., zero future deaths, the average of the last five years' estimated deaths, or equal to the population sustainability threshold (PST) assuming alternative values of ϕ). Population simulations were obtained using the 'Yearblock15' population model.

Future squid fishery deaths	λ_{2030} Decreasing	Stable	Increasing
Median historical deaths			
zero	0.959 (0.955–0.963)	0.989 (0.985–0.992)	1.060 (1.044–1.074)
average	0.958 (0.954–0.962)	0.988 (0.984–0.991)	1.060 (1.044–1.074)
$\phi = 0.05$	0.953 (0.948–0.957)	0.984 (0.979–0.988)	1.057 (1.041–1.072)
$\phi = 0.10$	0.946 (0.941–0.950)	0.978 (0.973–0.983)	1.054 (1.037–1.069)
$\phi = 0.20$	0.928 (0.922–0.934)	0.965 (0.959–0.971)	1.047 (1.030–1.063)
Upper 95% CI historical deaths			
zero	0.961 (0.957–0.965)	0.990 (0.986–0.994)	1.064 (1.049–1.076)
average	0.958 (0.954–0.962)	0.988 (0.983–0.992)	1.063 (1.048–1.075)
$\phi = 0.05$	0.955 (0.950–0.959)	0.985 (0.980–0.989)	1.061 (1.046–1.074)
$\phi = 0.10$	0.947 (0.943–0.952)	0.979 (0.974–0.984)	1.058 (1.043–1.071)
$\phi = 0.20$	0.930 (0.925–0.936)	0.966 (0.960–0.972)	1.052 (1.036–1.065)

Table A5-3: Projected population size of female New Zealand sea lions at the Auckland Islands in <u>2025</u> expressed as a percentage of projected population size under the zero future squid fishery (SQU 6T) related deaths scenario (N_{2025} % no squid fishery deaths), with alternative scenarios of: population growth (i.e., decreasing, stable or increasing); historical deaths relating to the (median or upper 95% of spatial risk model estimates); and future deaths relating to this fishery (i.e., the average of the last five years' estimated deaths, or equal to the population sustainability threshold (PST) assuming alternative values of ϕ). Population simulations were obtained using the '<u>Yearblock8</u>' population model.

Future squid fishery deaths	N_{2025} % no squid fishery deaths		
	Decreasing	Stable	Increasing
Median historical deaths			
average	99.5 (99.4–99.5)	99.5 (99.5–99.5)	99.6 (99.6–99.6)
$\phi = 0.05$	97.5 (97.4–97.6)	97.7 (97.6–97.8)	98.0 (97.9–98.1)
$\phi = 0.10$	95.0 (94.7–95.2)	95.4 (95.1–95.6)	96.1 (95.8–96.3)
$\phi = 0.20$	90.0 (89.5–90.5)	90.7 (90.2–91.2)	92.1 (91.7–92.5)
Upper 95% CI historical deaths			
average	99.0 (98.9–99.0)	99.0 (99.0–99.1)	99.2 (99.1–99.2)
$\phi = 0.05$	97.5 (97.4–97.6)	97.7 (97.6–97.8)	98.0 (97.9–98.2)
$\phi = 0.10$	95.0 (94.8–95.3)	95.4 (95.1–95.6)	96.1 (95.9–96.3)
$\phi = 0.20$	90.1 (89.6–90.6)	90.8 (90.3–91.3)	92.2 (91.8–92.6)

Table A5-4: Projected population size of female New Zealand sea lions at the Auckland Islands in <u>2025</u> expressed as a percentage of projected population size under the zero future squid fishery (SQU 6T) related deaths scenario (N_{2025} % no squid fishery deaths), with alternative scenarios of: population growth (i.e., decreasing, stable or increasing); historical deaths relating to the (median or upper 95% of spatial risk model estimates); and future deaths relating to this fishery (i.e., the average of the last five years' estimated deaths, or equal to the population sustainability threshold (PST) assuming alternative values of ϕ). Population simulations were obtained using the '<u>Yearblock15</u>' population model.

Future squid fishery deaths	N_{2025} as % of $N_{2025 \text{ no fishery deaths}}$		
	Decreasing	Stable	Increasing
Median historical deaths			
average	99.5 (99.4–99.5)	99.5 (99.5–99.5)	99.6 (99.5–99.6)
$\phi = 0.05$	97.5 (97.4–97.6)	97.7 (97.6–97.8)	97.9 (97.8–98.0)
$\phi = 0.10$	95.0 (94.8–95.3)	95.4 (95.1–95.6)	95.8 (95.6–96.0)
$\phi = 0.20$	90.1 (89.5–90.5)	90.7 (90.2–91.2)	91.7 (91.2–92.1)
Upper 95% CI historical deaths			
average	99.0 (98.9–99.0)	99.0 (99.0–99.1)	99.1 (99.1–99.2)
$\phi = 0.05$	97.5 (97.4–97.6)	97.7 (97.6–97.8)	97.9 (97.8–98.0)
$\phi = 0.10$	95.0 (94.8–95.3)	95.4 (95.1–95.6)	95.9 (95.6–96.1)
$\phi = 0.20$	90.1 (89.6–90.6)	90.8 (90.2–91.3)	91.7 (91.3–92.1)

Table A5-5: Projected population size of female New Zealand sea lions at the Auckland Islands in <u>2030</u> expressed as a percentage of projected population size under the zero future squid fishery (SQU 6T) related deaths scenario (N_{2025} % no squid fishery deaths), with alternative scenarios of: population growth (i.e., decreasing, stable or increasing); historical deaths relating to the (median or upper 95% of spatial risk model estimates); and future deaths relating to this fishery (i.e., the average of the last five years' estimated deaths, or equal to the population sustainability threshold (PST) assuming alternative values of ϕ). Population simulations were obtained using the '<u>Yearblock8</u>' population model.

Future squid fishery deaths	N_{2030} as % of $N_{2030\ \text{no fishery deaths}}$		
	Decreasing	Stable	Increasing
Median historical deaths			
average	98.8 (98.8–98.9)	99.0 (98.9–99.1)	99.3 (99.3–99.4)
$\phi = 0.05$	94.4 (94.1–94.7)	95.2 (94.9–95.5)	96.9 (96.6–97.1)
$\phi = 0.10$	88.8 (88.2–89.4)	90.5 (89.9–91.1)	93.7 (93.2–94.2)
$\phi = 0.20$	77.7 (76.5–78.8)	81.0 (79.7–82.1)	87.4 (86.3–88.4)
Upper 95% CI historical deaths			
average	97.7 (97.6–97.9)	98.1 (97.9–98.2)	98.7 (98.6–98.8)
$\phi = 0.05$	94.5 (94.2–94.8)	95.3 (95.0–95.6)	96.9 (96.7–97.2)
$\phi = 0.10$	89.0 (88.4–89.6)	90.6 (90.0–91.2)	93.8 (93.3–94.3)
$\phi = 0.20$	78.0 (76.9–79.3)	81.1 (79.9–82.3)	87.7 (86.6–88.6)

Table A5–6: Projected population size of female New Zealand sea lions at the Auckland Islands in <u>2030</u> expressed as a percentage of projected population size under the zero future squid fishery (SQU 6T) related deaths scenario (N_{2025} % no squid fishery deaths), with alternative scenarios of: population growth (i.e., decreasing, stable or increasing); historical deaths relating to the (median or upper 95% of spatial risk model estimates); and future deaths relating to this fishery (i.e., the average of the last five years' estimated deaths, or equal to the population sustainability threshold (PST) assuming alternative values of ϕ). Simulations were obtained using the '<u>Yearblock15</u>' population model.

Future squid fishery deaths	N_{2030} as % of $N_{2030\ \text{no fishery deaths}}$		
	Decreasing	Stable	Increasing
Median historical deaths			
average	98.8 (98.8–98.9)	99.0 (98.9–99.1)	99.3 (99.2–99.3)
$\phi = 0.05$	94.5 (94.2–94.8)	95.3 (95.0–95.5)	96.5 (96.2–96.7)
$\phi = 0.10$	89.0 (88.4–89.6)	90.5 (90.0–91.1)	93.0 (92.4–93.5)
$\phi = 0.20$	78.0 (76.8–79.1)	81.1 (79.9–82.1)	86.0 (84.8–87.0)
Upper 95% CI historical deaths			
average	97.7 (97.6–97.9)	98.1 (97.9–98.2)	98.6 (98.5–98.7)
$\phi = 0.05$	94.6 (94.3–94.9)	95.3 (95.0–95.6)	96.6 (96.3–96.8)
$\phi = 0.10$	89.1 (88.5–89.7)	90.6 (90.0–91.2)	93.1 (92.6–93.5)
$\phi = 0.20$	78.2 (77.0–79.4)	81.2 (79.9–82.3)	86.2 (85.1–87.1)