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Spatial assessment of fisheries risk for New Zealand sea lions at the Auckland Islands

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K. Large,
J. Roberts,
M. Francis,
D. Webber

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A spatially-explicit fisheries risk assessment (SEFRA) model was developed for female New Zealand sea lions around the Auckland Islands. Model inputs included: fishery data (commercial fisheries effort, observed effort and observer-reported sea lion captures), the estimated at-sea spatial density of female New Zealand sea lions around the Auckland Islands, and estimates of female population size through time (obtained from separate research).

This research collated and processed all available spatial tracking data of New Zealand sea lions monitored at the main breeding rookeries of the Auckland Islands, between 1996 and 2012. This comprised a total sample of 143 tracks of sufficient duration, of which 124 were from females. The spatial density of female New Zealand sea lions was estimated using a simple habitat model, fitted to the outputs of a hierarchical Switching State Space Model (SSSM) using groomed and filtered satellite fixes.

Key parameters estimated by the SEFRA model included the catchability of New Zealand sea lions by fishery group, the probability that an individual was alive given that it was captured, the probability that a capture was observable, and the number of adult females in the population each year. These parameters were then used to estimate annual deaths, population sustainability thresholds (PSTs) and risk ratios, using the standard SEFRA method, and assuming a base case calibration factor (ϕ) of 0.1. Future anthropogenic deaths at this level would result in suppression of the New Zealand sea lion population to 95% of unimpacted status under the assumption of logistic population growth.

Cryptic mortality rates associated with the use of sea lion exclusion devices (SLEDs) by squid trawls were estimated by separate research and used to estimate priors for the probability of observing capture events. For the squid fishery, the SEFRA model simulated from these priors to estimate deaths given estimated captures.

Risk model estimates of annual deaths were higher for scampi trawls (SCI 6A) (3.00 deaths yr^{-1}, 95% credible interval or CI = 1.00–6.33) than squid trawls (SQU 6T) (2.33 individuals yr^{-1}, 95% CI = 0.67–5.00) and all other Auckland Islands trawl fisheries combined (0.33 individuals yr^{-1}, 95% CI = 0.00–1.33) across the three latest years of the assessment (2014/15 to 2016/17). For the base case risk model, the annual risk ratio across all Auckland Islands trawl fishery groups was 0.24 (95% CI = 0.11–0.47) for this time period. Model runs using sensitivity ϕ values of 0.05 and 0.20 resulted in doubling and halving of risk ratios, respectively. These results indicate that, in the absence of other anthropogenic threats, annual commercial trawl deaths are unlikely to have been sufficient to suppress the Auckland Islands population of New Zealand sea lions below 90% of unimpacted status across the period 1992/93 to 2016/17. However, squid trawl effort at the Auckland Islands is known to have been greater in the 1980s, prior to the period addressed by this assessment.

Future research could extend this SEFRA model to include fishery records prior to 1992/93, including the 1980s period of relatively high squid fishery effort. Additional tracking of individuals of the large Dundas Islands population (including juveniles) would provide a tracking data sample that is more representative of the demographic composition of New Zealand sea lions at the Auckland Islands. Additional monitoring of New Zealand sea lion foraging in winter would provide the information requirements of a seasonal model, which may produce more robust estimates of risk for the scampi fishery. Further recommendations for future research are made at the end of the report text.

1. INTRODUCTION

The New Zealand sea lion has an extremely concentrated breeding distribution with 97–98% of annual pup production at the Auckland Islands and Campbell Island, in the New Zealand Sub-Antarctic region (Weir et al. 2019; Fyfe et al. 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 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 as "Nationally Vulnerable", the second 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) (Figure 1), *Klebsiella pneumonia* infection-related mortality of pups, and indications of variable and/or limited prey availability and consequent nutritional stress (Augé 2010; Meyer et al. 2015; Roe et al. 2015; Roberts & Doonan 2017; Roberts et al. 2018; Stewart-Sinclair 2013).

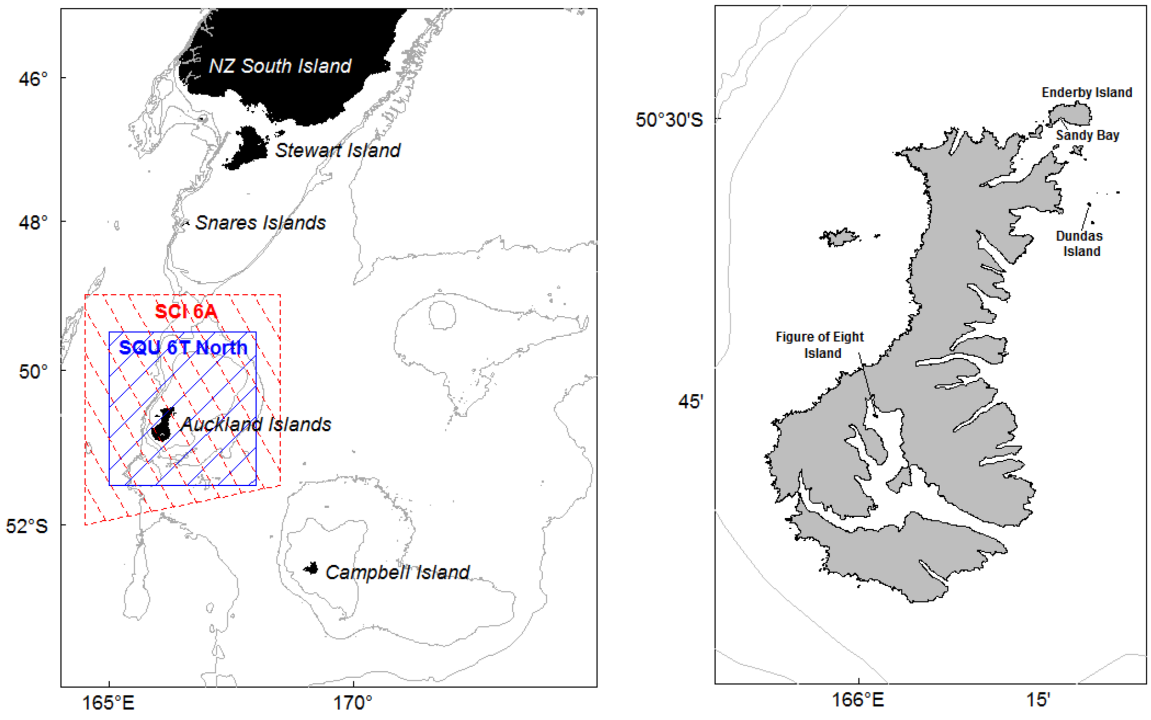


Figure 1: (left) Location of New Zealand sea lion breeding populations, including the Auckland Islands population addressed by this assessment. Grey lines represent the 200 m, 500 m and 1000 m bathymetric contours; the Northern Box of SQU 6T, where the Auckland Islands southern arrow squid fishery operates, is highlighted with a blue hatched polygon; the Auckland Islands scampi fishery area is highlighted with a red dashed hatched polygon (SCI 6A). (right) Map of the Auckland Islands showing the locations of the three New Zealand sea lion colonies where animals were tagged: Sandy Bay (on Enderby Island), Dundas Island and Figure of Eight Island.

Previous assessments estimating fisheries-related deaths of the Auckland Islands population of New Zealand sea lions (e.g. Abraham et al. 2017; Thompson et al. 2013) have been hampered by:

- A lack of pre-existing estimates of at-sea spatial density of New Zealand sea lions; and
- Uncertainty with respect to the cryptic mortality rate relating to the use of sea lion exclusion devices (SLEDs) in the squid trawl fishery at the Auckland Islands.

The historical satellite-derived spatial foraging distribution data were recently collated by DOC, including the data collected from more than 100 females using the main Auckland Islands breeding rookeries (Sandy Bay, Enderby Island; Dundas Island; and Figure of Eight Island) (Figure 1). In addition, new research has estimated cryptic mortality rates relating to the use of SLEDs in the squid fishery which can be used to estimate deaths given observed captures (Meyer 2019).

The spatially explicit fisheries risk assessment (SEFRA) approach (Sharp 2018) was used to estimate historical annual deaths and risk ratios of New Zealand sea lions for all commercial trawl fisheries around the Auckland Islands. This assessment used the newly available information to estimate the spatial density of New Zealand sea lions at the Auckland Islands. Capture rate in trawl fisheries is then estimated as a function of spatial overlap, using fisheries observer data. Separately (Meyer 2019) the latest information with respect to cryptic mortality rate is used to obtain robust estimates of annual deaths and population risk.

The specific research objectives of this research were to:

1. Characterise the foraging behaviour of Auckland Islands sea lions in a spatially and temporally explicit manner using available satellite telemetry data;
2. Apply spatial overlap methods to inform improved estimation of encounter rate, interaction rate, and cryptic mortality rate of Auckland Island sea lions with commercial fisheries over time, including for fishing effort with and without the use of SLEDs; and
3. Apply estimates from Objective 2 (with uncertainty) to inform spatially explicit estimates of fishery-related deaths in association with current fishing effort patterns.

All of these objectives are addressed by the research described by this report.

2. METHODS

2.1 Risk assessment methodology

This assessment used the SEFRA approach (Sharp 2018) to develop a spatially explicit fisheries risk assessment for female New Zealand sea lions at the Auckland Islands. The SEFRA method is briefly described in the next sub-section with detailed methods in Section 2.4.

2.1.1 SEFRA method

The spatial risk model was based on the SEFRA method (Sharp 2018), in which risk is expressed as a ratio between an estimate of fisheries-related deaths in the numerator and a ‘Population Sustainability Threshold’ (PST) in the denominator. The rate at which animals encounter the threat is estimated as a function of the spatial overlap between the threat intensity and the animal density in space, and the catchability (probability of capture per encounter) is estimated from observed capture events. Cryptic mortality rate (total deaths per observable capture) is estimated separately (Meyer 2019). The SEFRA method was deemed by an independent review initiated by Ministry for Primary Industries (now Fisheries New Zealand) to be a high-quality tool for spatial risk assessment (Lonergan et al. 2017).

The SEFRA method estimates annual fishery captures, derived annual mortalities, and relates this to a mortality threshold (PST)—the maximum number of annual deaths that a population unit can sustain without impacting on a population recovery objective. The SEFRA equation for estimating the Population Sustainability Threshold (PST) is

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

where ϕ is a calibration coefficient that can be tuned to achieve user-specified population-based management goals and account for alternative assumptions of the shape of population growth in response to density dependence; and N is the estimate of total population size.

The estimation of annual deaths (D) by SEFRA models (referred to as "Annual Potential Fatalities" or "APF" by previous SEFRA implementations, e.g., Abraham et al. 2017) is spatially-explicit, i.e., it accounts for spatial overlap when estimating deaths from information on capture rate. This is desirable when the spatial distribution of total fishing effort and the observed portion of that effort have a different degree of overlap with the assessed species, as may occur when the level of observer coverage is low.

The risk ratio (R) is then calculated as

$$R = \frac{D}{PST}$$

where R expresses fishery deaths (D) as a proportion of the threshold (PST) and is presented as a posterior probability distribution, propagating uncertainty in both D and the PST.

2.1.2 SEFRA inputs

Detailed methods for estimating annual threat-specific deaths (D) are given in Section 2.1. This calculation requires information with respect to:

- The spatial distribution of New Zealand sea lions;
- Annual estimates of total female population size;
- Spatially resolved commercial fisheries observer records of captures; and
- Spatially resolved records of all commercial fishing effort records (i.e., including observed and unobserved events), so that capture rate information from above can be used to estimate the total number of captures or deaths relating to a threat given spatial overlap.

The derivation of PST requires information with respect to:

- Intrinsic population growth rate ($r^{\max}$);
- A specified population reference outcome to inform the choice of the calibration coefficient (ϕ). The reference outcome is expressed in terms of recovery to and/or stabilisation of the impacted population at a defined proportion of the unimpacted population state, at equilibrium; and
- Annual estimates of total female population size (note that this is also used for estimating D).

All SEFRA inputs were estimated/updated by the commercial fisheries risk assessment for Auckland Islands New Zealand sea lions, with the exception of: intrinsic population growth rate ($r^{\max}$); annual female population size (N_y), for which the recent approved estimates were used (Roberts 2017); and the calibration coefficient (ϕ), for which values specified by Fisheries New Zealand were used (see below).

2.2 Processing sea lion track data

The spatial distribution of individual New Zealand sea lions has previously been tracked around the Auckland Islands using electronic devices (e.g., Chilvers et al. 2011). Track locations were determined using electronic transmitters (hereafter called tags) that communicate with orbiting *Argos* satellites, and the depths of the animals in the water column were recorded with Time-Depth Recorders (TDRs). Details of instruments used and tagging procedures are given in a number of other studies (Gales & Mattlin 1997; Chilvers et al. 2005; Leung et al. 2013) and are summarised briefly below.

2.2.1 Metadata

We were unable to access all known tag and TDR datasets from New Zealand sea lions at the Auckland Islands, or a comprehensive metadata description for the tagged animals. We therefore inspected a range of sources for factors crucial to our study: tagging location (Sandy Bay, Dundas Island, or Figure of Eight Island; see Figure 1), season (Summer or Winter), sex (Male or Female), and demographic stage (Yearling, Juvenile, Lactating female, Adult male). Sources searched for metadata included published papers, Microsoft Excel spreadsheets provided by DOC and scientists connected with the tagging programmes, field notebooks and other field records.

Some tag datasets were not available for this study. Tag datasets mentioned in published and unpublished documents that we did not access included four tags from 2001, one tag from 2002, five tags from 2005 and 17 tags from 2011. Frequently, we could not match tag datasets with individual sea lions reported in published papers, because the latter often referred to animals using an identifier or a brand number rather than the tag number. In particular, most of the tag data for 2005 to 2008 could not be linked to individual animals. Fortunately, however, our analyses focused on demographic groups of animals rather than individuals, and it was therefore only necessary to determine the tagging location, season, sex and demographic stage for each tag dataset. Season (i.e., summer or winter) was determined from the date-time stamp in the dataset. Other factors could usually be inferred because tagging tended to focus on a specific demographic group and tagging location each year and season (e.g., lactating females tagged at Sandy Bay). There was sufficient information to classify all tag datasets except for two GPS tags deployed on females at Sandy Bay in 2010, which could have been either juvenile or lactating females.

2.2.2 Tag data grooming and processing

Tags and TDRs were mounted on a square of neoprene rubber which was then attached to the dorsal fur of anaesthetised sea lions with an epoxy cement. Tags were usually removed at the end of the field season when sea lions returned to the rookeries. In a few cases, animals were not resighted and their tags were not recovered; these tag packages would have been shed when the sea lion subsequently moulted.

In the early years of the programme, Telonics 300 tags (models ST6 and ST10) were used (Gales & Mattlin 1997; Chilvers et al. 2005). These tags were also used in subsequent years, and supplemented with more modern Wildlife Computers Splash tags (Leung et al. 2013). Currently we have insufficient information to determine the tag brand and model for most *Argos* datasets after the 1990s. In 2010–2012, two or three GPS tags were deployed per year, sometimes in tandem with a satellite tag. The GPS tags were most likely to have been Sirtrack F1G138B FASTLOC models, as used on New Zealand sea lions at nearby Stewart Island (Chilvers 2018).

Most tags were equipped with wet/dry sensors and automatically stopped recording and transmitting data when the sea lions were on land. Tags used in the early 1990s did not have wet/dry sensors. However, scientists recorded the times of sea lion departure from and return to the breeding colony, and manually identified dataset records as 'wet' or 'dry'. In the present study we omitted all 'dry' records from the data.

An estimate of location error is provided by the *Argos* satellite system for all location fixes. Fixes are classified into seven 'Location Classes': G (GPS tags), 3, 2, 1, 0, A, B and Z in descending order of accuracy (Table 1). Class Z fixes are invalid locations and were removed from the dataset. Class A and B fixes are not assigned an accuracy estimate by *Argos*. Independent studies indicate that the *Argos* estimates of accuracy are overly optimistic, and that Classes 3, 2, 1 and A are accurate to about 2 km and Classes 0 and B are accurate to about 5–10 km (Boyd & Brightsmith 2013). GPS tags are stated by *Argos* to be accurate to 20–70 m (Table 1).

Table 1: Estimated *Argos* satellite fix accuracy (Boyd & Brightsmith 2013).

Argos location class	Stated accuracy (metres)	Locations received per satellite pass
3	< 250	≥ 4
2	250–500	≥ 4
1	500–1000	≥ 4
0	> 1500	≥ 4
A	Not stated	3
B	Not stated	1–2
Z	No location	0
Fastloc GPS	20–70	≥ 1

Raw *Argos* datasets were not available for sea lions tagged in 1995 and 1996, and we were limited to using filtered datasets from which Class A and B fixes had been removed. For all other years, the tag datasets included the full range of classes.

Multiple levels of data grooming were applied to the combined datasets (N = 41 341) to remove the following probable errors:

1. Extreme outliers (north of 42 °S latitude and outside 145–185 °E longitude) were removed (N = 182).
2. Additional outliers at the start and end of datasets, or before or after a haul-out period (i.e., either side of a data gap of several days), were removed (N = 118). These outliers, which were identified manually by inspecting tracks fitted to the raw fixes, were not removed by the automated speed filter (see the next grooming step). The speed filter analyses five-point sequences of fixes and is therefore unable to identify probable errors in the first two or last two points of a dataset, or either side of a haul-out period. Some short sequences of fixes that followed data gaps of seven or more days at the ends of datasets were also removed.
3. A speed filter was applied to the remaining *Argos* data (excluding GPS data which were all retained) using the *argosfilter* package in the open-source statistical programming language *R* (Freitas et al. 2008; R Development Core Team 2017). This filter removed fixes that required a swimming speed between fixes of greater than 3.5 m.s^{-1} (302 km.day^{-1}), unless they were within 5 km of the previous position. This latter constraint prevents removal of locations that generate artificially high-speed estimates as a result of two fixes being obtained within a short time (Freitas et al. 2008). The speed threshold was based on frequency distributions of active swimming speeds of 12 Sandy Bay sea lions which showed that they rarely exceeded 3.0 m.s^{-1}. That value was then adjusted upwards by 15% to allow for the animal 'porpoising', during which the propeller of the speed meter stops rotating, and then rounded up (to 3.5 m.s^{-1}) (Crocker et al. 2001). The angle filter option of the *argosfilter* package was not used because it was found to considerably reduce the dataset size without producing any obvious improvement to the track that was not subsequently accounted for by fitting a movement model (see Section 3.1).

2.2.3 Sea lion tracks

A hierarchical Switching State Space Model (SSSM) was fitted to groomed and filtered *Argos* fixes to estimate equally-spaced locations and to classify movements into two behavioural states based on distance travelled and changes in course (Jonsen et al. 2007). One state, characterised by slow speeds and frequent changes of direction, was called Resident, and a second state, characterised by rapid movements over long distances with few or small direction changes, was called Travel (Jonsen et al. 2007). This modelling approach was specifically developed for use with gappy and error-prone satellite tracking data (Jonsen et al. 2005; 2007). The hierarchical model was fitted to data from all sea lion tracks simultaneously. Hierarchical models estimate a single set of movement parameters simultaneously for all animals, rather than separately for each animal, and this provides improved behavioural state estimation through reduction of uncertainty (Jonsen 2016). Errors were modelled with t-distributions to minimise the impact of erroneous locations, and different error distributions were allowed for each of the *Argos* location classes, thus accounting for variable location accuracy among classes (Jonsen et al. 2005; 2007). Two chains of 60 000 Markov Chain Monte Carlo (MCMC) samples were used, with the first 30 000 being discarded as the adaptation and burn-in phase. The remaining 30 000 samples were thinned to 1000 (every 30th sample) to minimise within-chain sample autocorrelation (Jonsen et al. 2007). Models were fitted using the *bsam* package in *R*, which in turn used JAGS 4.2.0 software to perform Bayesian inference (Jonsen et al. 2005; Plummer 2016)

The model time step was determined iteratively, starting at 0.1 days and increasing to 0.3 days, with the latter being selected as the best choice for this dataset. Short-time steps provide higher resolution tracks, but the frequency of satellite fixes was often not sufficient to warrant the higher resolution, particularly in the early years of the study when tag transmission success was low. Tests with time steps of 0.1 and 0.2 days resulted in frequent 'overshoots' in which fitted tracks extended beyond the data points, usually at or near a haul-out location and time.

Trials showed that the SSSM did not converge for datasets having fewer than 20 fixes. Fitted tracks for datasets with fewer than about 40 fixes showed unacceptable over-shoots. We therefore removed tracks that had 40 or fewer fixes from the analyses.

Values of the behavioural mode parameter *b* were used to assign a behavioural state for each sea lion at each fitted track location. *b* values can range from 1 (*Travel* state with high certainty) to 2 (*Resident* state with high certainty). In this study, *b* values less than 1.3 were interpreted as *Travel* and *b* values greater than 1.7 as *Resident*. Intermediate values of *b*, which indicate an uncertain behavioural state, were classified as *Undefined*. These classification criteria are subjective, and the time step of the fitted model averages the movement signal across a 0.3-day period, so the inferred *Resident* and *Travel* locations may not reflect the true behaviour of sea lions in those locations.

For display purposes, spatial probability distributions of sea lions were developed using Kernel Utilisation Distributions (KUDs) and the *R* package *adehabitatHR* (Worton 1989; Calenge 2006). This package was developed for small-scale, equal-area spatial grids, rather than a latitude/longitude coordinate system. Consequently, we converted our SSSM locations from latitude/longitude to Universal Transverse Mercator (UTM) locations centred on zone 58 (162–168 ^{o}E) using package *PBSmapping*, and then formatted them as *SpatialPoints* objects with package *sp*. KUDs were then estimated using the function *kernelUD* in *adehabitatHR* and a pre-defined spatial grid, and 50% and 80% probability contours were generated. The results were then converted back to latitude/longitude coordinates for plotting on maps.

2.3 Spatial density of New Zealand sea lions

This subsection describes the methods that were used to estimate the spatial density of female New Zealand sea lions from locations obtained using the SSSM (previous section). Predictive models were fitted to these data, without making a distinction with respect to state (i.e., foraging or travelling). Habitat models often exclude travelling locations from model fitting on the basis that they are less likely to correspond to actual habitat requirements. However, the objective of the modelling described here was to estimate the overall at-sea spatial density of female New Zealand sea lions and, so, no distinction was made between foraging and travelling states.

A true habitat modelling approach was initially trialled (not reported on here), which predicted the spatial density of New Zealand sea lions in response to habitat variables (e.g., incorporating chlorophyll *a* concentration and estimated prey species density), although this approach was constrained by a lack of prey species information at key, well-foraged locations. As such, an alternative approach was used, which related sea lion locations to the basic locational variables (longitude, latitude, depth and distance to colony), to estimate their spatial density around the Auckland Islands. Since these models use longitude and latitude variables, they have no predictive power for New Zealand sea lions foraging from other locations (e.g., from Campbell Island, or the New Zealand mainland).

Generalised additive models (GAMs) were developed using the *bam* function of R package *mgcv* (R Development Core Team 2017; Wood 2011; Wood et al. 2015). This implements GAM models optimised for very large datasets, whilst producing very similar predictions to standard GAMs, given the same model and basic structure (Wood et al. 2015). The response variable was the sum of each individual's locations aggregated by 1×1 km grid cell. This produced a grid of aggregated frequencies for each individual (including zeros in all cells for which a sea lion had no positive locations) for all 1×1 km cells with a specified area (from 165.592 °E to 168.174 °E, and 51.212 °S to 49.502 °S).

The GAM structure used for all New Zealand sea lion demographic groups was

$$\mathit{freq} \sim \mathrm{te}(\mathit{lat},\mathit{lon}) + \mathrm{s}(\mathit{dep}) + \mathrm{s}(\mathrm{dtc}) + \mathrm{s}(1|\mathit{dep}:\mathit{ID}) + \varepsilon$$

where *freq* was the total number of locations for each individual by 1×1 km grid cell; *lat*, *lon*, *dep*, and *dtc* were, respectively, the latitude, longitude, depth (Figure A2-1), and distance to colony (dependent on where the individual was instrumented) for each corresponding 1×1 km grid cell; and ε was the error term. A bivariate spline was specified for latitude and longitude, using a tensor product smoother. Univariate splines were specified for depth and distance to rookery using standard GAM spline smoothers (Wood 2017) with cubic regression splines (bs = "*cr*"), and the basis dimension constrained ($k = 4$) to prevent biologically implausible relationships. In short, models estimated the observed number of presences of a sea lion in each grid cell, conditional on a surface smoother of latitude and longitude and the relationship with depth and distance to colony.

A normal random slope was specified by individual sea lion for the depth variable (bs = "*re*"), relating the response to depth for each individual. This approach was taken to account for individual differences in satellite tag deployment duration (i.e., some individuals had many more reported locations than others) and strong individual foraging site fidelity of female New Zealand sea lions (Chilvers 2008), which would otherwise have overly represented the foraging characteristics of individuals with the most reported locations.

All GAMs assumed a negative binomial error structure (family = '*nb*'), with the shape parameter (θ) estimated during model fitting, to account for over-dispersion in the response variable. Quantile-quantile plots were produced for each model to check that distributional assumptions were met. Spatial plots of the coefficient of variation (CV) associated with model estimates were used to identify regions where predictive models were likely to be extrapolating beyond the modelled habitat envelope.

The spatial density predictions for different demographic groups (e.g., by breeding status and rookery) were then combined to produce a prediction for all females at the Auckland Islands. This was achieved by rescaling the spatial predictions by the relative population size of each group:

1. First, the spatial density raster for each demographic group was rescaled to sum to 1.
2. For lactating female groups (e.g., Sandy Bay, Dundas Island, Figure of Eight Island):
    a. Each raster was multiplied by the 2017/18 pup production estimate (the latest at the time of this analysis): Dundas Island = 1397 pups, Sandy Bay = 332 pups, Figure of Eight Island = 63 pups (DOC 2019).
    b. The three rasters were then summed to produce a combined density of lactating female New Zealand sea lions across all three breeding rookeries.

   c. The combined lactating female raster was then rescaled to sum to 1.
3. The juvenile spatial density raster was then divided by 3 and summed with the rescaled lactating female raster (which approximates the relative proportions of females at ages most vulnerable to bycatch (3+) that are juveniles (ages 3–5) (Breen et al. 2016)).
4. The combined juvenile/lactating female raster was then rescaled to sum to 1.

The rescaled raster was then used as an input to the spatial risk model.

2.4 Spatial fisheries risk assessment

The SEFRA method is used to estimate commercial fishery-related deaths for New Zealand sea lions. The conceptual and mathematical basis of the SEFRA method, including the format of data inputs and underlying structural assumptions, is described in Sharp (2018).

The following databases were provided by Fisheries New Zealand, on 6 August 2018:

1. Spatially resolved commercial fishing effort data, per fishing event;
2. Fisheries observer data, per fishing event;
3. Fisheries observer-recorded protected species captures, per capture event.

These databases, along with spatio-temporal species distribution maps and prior distributions for all model parameters, inform the SEFRA model. All variables used to describe the model can be found in Table **2**.

This implementation of the SEFRA method was not seasonal (i.e., the model was not partitioned into summer and winter periods like previous SEFRA models, e.g., Roberts et al. (2019)), due to a lack of sufficient sea lion foraging information in winter months (see below). Models only included commercial trawls, the only gear type known to pose a risk to New Zealand sea lions at the Auckland Islands. Only trawl fishing events in the Auckland Islands area were used (by taking a subset of fishing events for which trawl_area = “SQUAK6”). Fishing events were either observed (an observer was on board the fishing vessel at the time of a capture) or unobserved. New Zealand sea lion captures recorded by fisheries observers on the observed portion of the fishing effort were used to estimate model parameters by fitting a relationship between this effort and observed sea lion captures. Observed sea lion captures were recorded as being alive or dead. This characteristic is also reflected in the model and subsequently used to estimate the number of deaths, which assumes that not all animals captured alive and released will die. The combination of observed and unobserved effort, along with the estimated parameters, was used to estimate the total number of commercial fishery related captures, deaths, and risk.

Table 2: Variable symbols, support, and descriptions. Estimated parameters are estimated within the model while random variables are drawn from a prior distribution outside of the model (i.e., within the generated quantities block of the Stan code) (Stan Development Team, 2018).

Symbol	Support	Description
Indices		
i	$i = \{1,2,\ldots\}$	A fishing event (i.e., a trawl event) that occurs at a time and location
g	$g = \{1,2,\ldots,8\}$	Commercial fishery group (see Table **3**)
c	$c = \{1,2,\ldots\}$	A cell in a map
y	$i = \{1993,1994,\ldots,2017\}$	Model year (e.g., fishing season 1992/93 is labelled year 1993)
Data		
$\left(C_{yg}^{\text{live}}\right)'$	≥ 0	Number of observed live captures of females by year and fishery group
$\left(C_{yg}^{\text{dead}}\right)'$	≥ 0	Number of observed dead captures of females by year and fishery group
Covariates		
a'_{ygi}, a_{ygi}	≥ 0	Observed fishing intensity and fishing intensity (number of trawl tows) by year and fishery group
p_i	$\in (0,1)$	Relative spatial density of female New Zealand sea lions
O'_{yg}, O_{yg}	≥ 0	Observed overlap and overlap by year and fishery group

Estimated parameters		
v_g	≥ 0	Catchability by fishery group
p_g^{obs}	$\in (0,1)$	Probability that a capture event is observable by fishery group
ψ	$\in (0,1)$	Probability of an individual being alive given that it is caught
N_y	≥ 0	Annual female population size by year
Random variables		
r^{max}	> 0	Intrinsic population growth rate
ω	$\in (0,1)$	Live release survival rate
Fixed parameters		
ϕ	$\in (0,1)$	Calibration coefficient
Derived quantities		
$C_{yg}^{\text{live}}, C_{yg}^{\text{dead}}$	≥ 0	Live and dead captures of females by year and fishery group
D_{yg}	≥ 0	Annual deaths by fishery group
PST_y	≥ 0	Annual population sustainability threshold
R_{yg}	≥ 0	Annual risk ratio by fishery group
k_g	≥ 0	Cryptic mortality by fishery group

2.4.1 Estimating key model parameters

In consultation with the Fisheries New Zealand Aquatic Environment Working Group, a total of eight fishery groups (g) were used by the model, including: trawls targeting southern arrow squid (six groups for different trawl gear types and year blocks with different SLED use categories); trawls targeting scampi; or trawls targeting all other species (including deep water species, e.g., orange roughy (*Hoplostethus atlanticus*) and oreo species (*Oreosomatidae*); middle depth species, e.g., hoki (*Macruronus novaezelandiae*) and ling (*Genypterus blacoides*); and shallower distributed species, e.g., red cod (*Pseudophycis bachus*) and silver warehou (*Seriolella punctata*)) (Table **3**).

Table 3: Spatial risk model fishery groups

Fishery group	Description	Fishing years
$\text{SQU}_{\text{BT\_NO\_SLED}}$	Bottom trawls targeting southern arrow squid, not fitted with a sea lion exclusion device (SLED)	1992/93 to 2007/08
$\text{SQU}_{\text{BT\_ NONSTAND\_SLED}}$	Bottom trawls targeting southern arrow squid, fitted with a non-standardised SLED	2000/01 to 2007/08
$\text{SQU}_{\text{BT\_STAND\_SLED}}$	Bottom trawls targeting southern arrow squid, fitted with a standardised SLED	2008/09 to 2016/17
$\text{SQU}_{\text{MW\_ NO\_SLED}}$	Midwater trawls targeting southern arrow squid, not fitted with a SLED	1992/93 to 2007/08
$\text{SQU}_{\text{MW\_ NONSTAND\_SLED}}$	Midwater trawls targeting southern arrow squid, fitted with a non-standardised SLED	2000/01 to 2007/08
$\text{SQU}_{\text{MW\_STAND\_SLED}}$	Midwater trawls targeting southern arrow squid, fitted with a standardised SLED	2008/09 to 2016/17
SCI	Trawls targeting scampi	1992/93 to 2016/17
OTH	Trawls targeting all other species	1992/93 to 2016/17

Fishing effort is described per fishing event (i.e., not aggregated within cells). Every fishing event (i) has an associated fishing intensity (a_{ygi}) measured in the number of trawl events each fishing year (y) within a fishery group (g). Observed fishing events are denoted using the prime symbol as a'_{ygi} where

$$a'_{ygi} \subset a_{ygi}$$

meaning that observed effort is a subset of all fishing effort.

Estimated parameters included Catchability (i.e., the probability of capture or death) in each commercial fisheries group (v_g), the probability that an event is observable for each fisheries group (p_g^{obs}), the probability of an individual being alive (given that it is captured) for each fisheries group (ψ), and the live-release survival rate for each fisheries group (ω). Prior distributions for the annual population size estimates (N_y) are specified in the model. Catchability, the probability that an event is observable, the probability of an individual being alive given that it is captured, and the live release survival rate are parameters that are required to be estimated.

Fisheries observer data (protected species captures on observed fishing events) are used to estimate model parameters. Observed overlap is calculated for each year (y) and commercial fishery group (g) using data from 1992/93 to 2016/17

$$O'_{yg} = \sum_i a'_{ygi} p_i$$

The probability of an individual being alive given that it is captured (ψ) is then used to calculate the annual expected number of observed alive (λ^{live}) and dead (λ^{dead}) captures

$$\lambda^{\text{live}}_{yg} = O'_{yg} N_y v_g p^{\text{obs}}_g \psi$$

$$\lambda^{\text{dead}}_{yg} = O'_{yg} N_y v_g p^{\text{obs}}_g (1 - \psi)$$

Cryptic mortality rate is defined as

$$k_g = \frac{1}{p^{\text{obs}}_g}$$

2.4.2 Bayesian Inference

2.4.2.1 Prior distributions and simulated random variables

Prior distributions relate to estimated model parameters. Random variables are drawn from a distribution (in the generated quantities block in the Stan code) (Stan Development Team, 2018) and are therefore not updated by data.

Truncated normal prior distributions were assumed for the annual population size of female New Zealand sea lions at the Auckland Islands (N_y). The parameters used to specify the prior distributions were derived from the MCMC samples of annual female population size from the base case model developed by Roberts (2017). The annual population size prior parameters are shown in Table 4.

Table 4: Female annual population size normal prior parameters

Year	Mean (standard deviation)	Year	Mean (standard deviation)
1993	6 672 (368)	2006	5 821 (137)
1994	7 163 (347)	2007	5 601 (130)
1995	7 194 (303)	2008	5 202 (115)
1996	7 327 (265)	2009	4 783 (100)
1997	7 477 (230)	2010	4 824 (100)
1998	7 625 (201)	2011	4 641 (100)
1999	7 635 (198)	2012	4 513 (97)
2000	7 422 (180)	2013	4 532 (110)
2001	7 208 (166)	2014	4 228 (140)
2002	6 990 (155)	2015	4 102 (193)
2003	6 775 (148)	2016	4 136 (294)
2004	6 557 (145)	2017	4 333 (361)
2005	6 318 (145)		

Uninformative Catchability priors were specified in log-space to help with MCMC mixing

$$\log(v_g) \sim \mathrm{normal}(0, 10^2)$$

For all trawls not targeting squid, and trawls targeting squid without a SLED ($\mathrm{SQU}_{\mathrm{BT\ NO\_SLED}}$ and $\mathrm{SQU}_{\mathrm{MW\ NO\_SLED}}$), all capture events were assumed to be observable ($p^{\mathrm{obs}} = 1$) (i.e., there was no cryptic mortality, so that estimated deaths equal captures). Catchability was estimated separately for squid trawls using non-standardised SLEDs ($\mathrm{SQU}_{\mathrm{BT\ NONSTAND\ SLED}}$ and $\mathrm{SQU}_{\mathrm{MW\ NONSTAND\ SLED}}$) in years 2001 to 2008. For squid trawls from 2009 on, the prior distribution for the probability that an event is observable was obtained from a separate project estimating New Zealand sea lion cryptic mortality rates in squid trawls (Meyer 2019). Informative prior distributions were assumed for captures in midwater trawls targeting southern arrow squid since 2009

$$p^{\mathrm{obs}}_{g=SQUMW} \sim \mathrm{beta}(10.617, 5.670)$$

and for squid bottom trawls since 2009

$$p^{\mathrm{obs}}_{g=SQUBT} \sim \mathrm{beta}(39.225, 5.885)$$

A vaguely informative prior was used for the probability of an individual being alive

$$\psi \sim \mathrm{beta}(1, 3)$$

The live-release survival rate random variable used the prior specified for a SEFRA model of Hector's and Māui dolphins (Roberts et al. 2019), and was drawn from

$$\omega \sim \mathrm{uniform}(0.5, 0.9)$$

A normal prior was assumed for the intrinsic population growth rate ($r^{\max}$) random variable

$$r^{\max} \sim \mathrm{normal}(0.12, 0.01^2)$$

where the mean $r^{\max}$ of 0.12 approximates to the corresponding $R^{\max}$ value of 0.12 typically used as the default value used for the population assessment of pinnipeds (e.g., Wade 1998); the standard deviation value of 0.01 is consistent with the $r^{\max}$ 95% CI = 0.10–0.14.

A summary of all priors used in the SEFRA model is given in Table 5.

Table 5: Table of prior distributions for the model parameters and the generated random variables

Parameter	**Symbol**	**Subscript**	**Prior type**	**Prior parameters**	
Catchability	$\log(v_g)$	$\forall g$	normal	$\mu = 0$	$\sigma = 10$
Probability capture event is observable	p^{obs}_g	$g = SQUMW$	beta	$\alpha = 10.617$	$\beta = 5.670$
		$g = SQUBT$	beta	$\alpha = 39.225$	$\beta = 5.885$
Probability an individual is alive when captured	ψ	-	beta	$\alpha = 1$	$\beta = 3$
Female annual population size	N_y	$y = \{1993, 1994 \ldots\}$	normal	See Table 4.	
Live-release survival probability	ω	-	uniform	0.5	0.9
Intrinsic population growth rate	r^{max}	-	normal	$\mu = 0.12$	$\sigma = 0.01$

2.4.2.1 Likelihood

A Poisson distribution was assumed for the observable alive and dead captures

$$\left(C_{yg}^{\text{live}}\right)' \sim \text{Poisson}\left(\lambda_{yg}^{\text{live}}\right)$$
$$\left(C_{yg}^{\text{dead}}\right)' \sim \text{Poisson}\left(\lambda_{yg}^{\text{dead}}\right)$$

noting that the Poisson distribution has the same mean and variance.

Bayesian inference was done using *Stan*, making use of its Hamiltonian Monte Carlo (HMC) algorithm (Stan Development Team, 2018). Four MCMC chains were generated. Each chain was run for 200 000 iterations, the first 50 000 iterations were used during the warm-up phase and were discarded. Every 100th iteration was kept after warm-up. This resulted in a total of 1500 samples for each chain, and a total of 6000 samples used for the posterior distribution of the model.

Model predictions were obtained using both observed and unobserved fishing events (a_{ygi}). Overlap (observed and unobserved fishing events) was calculated by year (y) and fishery group (g)

$$O_{ygi} = a_{ygi} p_i$$

A live-release survival rate random-variable (ω) was used to calculate the number of deaths by year and fishery group

$$D_{ygi} = O_{ygi} N_y v_g (1 - \psi\omega)$$

A mortality constraint (whereby total deaths are not allowed to exceed 1 minus the annual non-calf survival rate), as described in Sharp (2018), was not used because annual survival estimates were not available to relate to annual estimated deaths for years prior to 2000.

The annual population sustainability threshold (PST) was calculated as

$$PST_y = \frac{1}{2} \phi r^{max} N_y$$

where ϕ is the calibration coefficient with a base case value of $\phi = 0.10$ and sensitivity values of $\phi = 0.05$ or $\phi = 0.20$ specified by Fisheries New Zealand. These values of ϕ are consistent with population recovery to 95%, 97.5% or 90%, respectively, of unimpacted population size when assuming logistic population growth. The annual risk ratio for each fishery year and group (R_{yg}) is

$$R_{yg} = \frac{\sum D_{ygi}}{PST_y}$$

where a risk ratio > 1 is consistent with annual deaths for the respective fishery year and group exceeding the annual PST.

In addition, to approximate risk given 'current' female New Zealand sea lion population size at the Auckland Islands, and intensity of fishing effort, we used the most recent three years of data (2014/15 to 2016/17), and estimate the mean risk across these three years for each fishery group

$$R_g = \left(\sum_y \frac{\sum_{i \in yg} D_{ygi}}{PST_y} \right) \times \frac{1}{n}$$

where n is the number of years that we are averaging across ($n = 3$).

And the combined risk across all fishery groups is

$$R = \left(\sum_y \frac{\sum_{ig} D_{ygi}}{PST_y} \right) \times \frac{1}{n}$$

where n is the number of years that we are averaging across ($n = 3$).

3. RESULTS

3.1 New Zealand sea lion track data

The ungroomed tag dataset comprised 41 341 position fixes, which was reduced to 40 524 fixes after the removal of two satellite tag datasets from sea lions that were also tagged with GPS tags, and then was reduced further to 40 224 fixes after the first two data grooming steps (Section 2.2.2). The automated speed filter (the third grooming step) removed 15.2% of the remaining fixes (Figure A1-1), resulting in a final dataset of 34 099 fixes, or 84.1% of the original data. The speed filter retained a high percentage (96–99%) of fixes in location classes 1–3 and retained considerably smaller percentages of fixes in classes 0, A, and B (74–80%) (Figure A1-1). The filtered dataset was greatly improved with all obvious outliers removed (Figure 2). The final dataset was dominated by fixes with intermediate accuracy (49% location classes 1 and 0) (Figure A1-2). High-accuracy fixes (location classes GPS, 3 and 2) made up about one-quarter (24%) of fixes.

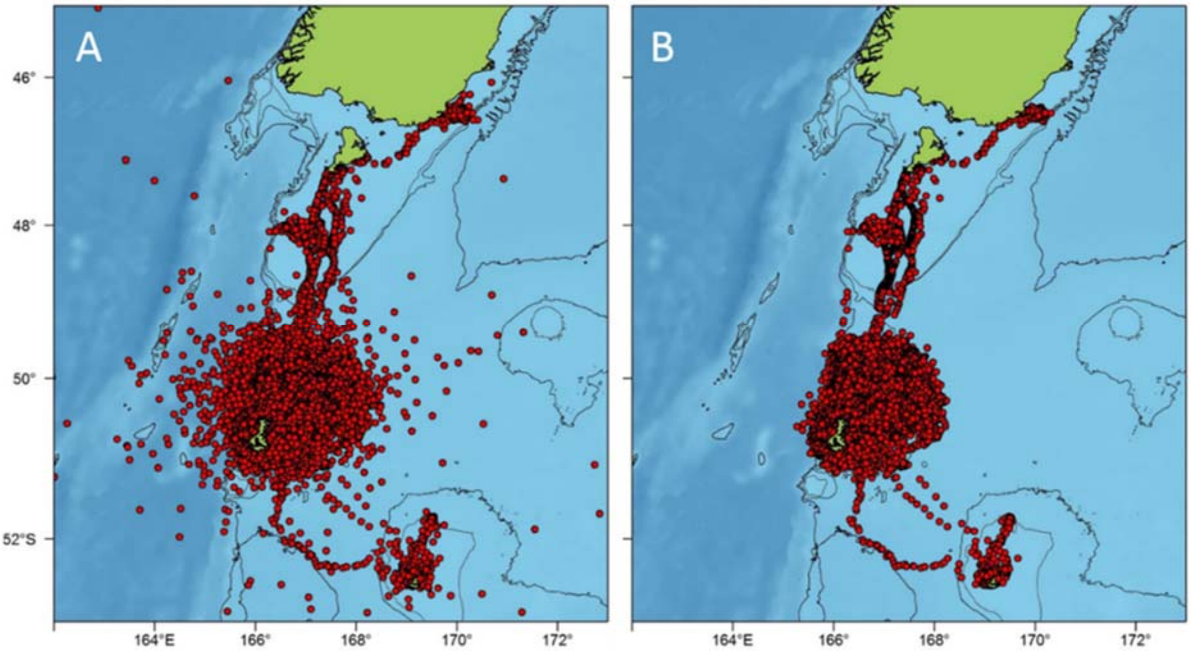


Figure 2: Comparison of A) unfiltered fixes and B) speed-filtered fixes for New Zealand sea lions. Isobaths are at 250, 500 and 1000 m.

There was a marked improvement in tag technology over the course of the study. In the 1990s, satellite tags almost always produced fewer than 10 fixes per day at sea, and that increased to 10–20 fixes per day in the 2000s (Figure A1-3). GPS tags were highly variable, but produced 30–80 fixes per day.

The filtered dataset comprised 165 tag 'tracks', where a track is defined as a unique tag-sea lion combination in a given season-year temporal stratum. A track comprised all of the foraging trips by a sea lion away from the colony within a season. To fit the SSSM, tracks with 40 or fewer fixes were omitted, leaving 143 tracks and 33 751 fixes for analysis.

The SSSM dataset is summarised in Table 6. Most tag tracks (72%) were for lactating females, followed by juveniles of both sexes (22%). Few yearlings (*N*=5) and adult males (*N*=1) provided useable tracks. Based on their sizes, the two Sandy Bay females with unknown demographic state were either juveniles or lactating adults.

Table 6: Count of individuals by location, sex and demographic status from which telemetry data were collected and used for fitting a SSSM model.

Demographic status	**Sandy Bay** **Female**	 **Male**	**Dundas Island** **Female**	 **Male**	**Figure of Eight Island** **Female**	 **Male**
Yearling	2	3	0	0	0	0
Juvenile	17	15	0	0	0	0
Lactating	75	-	24	-	4	-
Adult (unknown breeding status)	2	1	0	0	0	0
Total	96	19	24	0	4	0

At Sandy Bay, lactating females were tagged in most years between 1996 and 2012. Tagging at the other two Auckland Islands rookeries was much more restricted: lactating females were tagged at Dundas Island only in 2005–2007, and at Figure of Eight Island only in 2008 (Table 7). All juvenile sea lions were tagged at Sandy Bay in 2007–2010. Most of the filtered fixes (95.0%) occurred in Summer, particularly in January–February (Figure 3). Only 5.0% of fixes occurred in Winter, and all of them were in 1996 and 1997.

Table 7: Count of lactating females by location and year from which telemetry data were collected and used for fitting a SSSM.

	Lactating females from which tracking data were obtained, by Auckland Islands rookery		
Year	**Sandy Bay**	**Dundas Island**	**Figure of Eight Island**
1996	6	0	0
1997	7	0	0
1998	5	0	0
1999	0	0	0
2000	3	0	0
2001	0	0	0
2002	5	0	0
2003	7	0	0
2004	10	0	0
2005	8	7	0
2006	2	8	0
2007	0	9	0
2008	2	0	4
2009	2	0	0
2010	5	0	0
2011	2	0	0
2012	11	0	0
Total	75	24	4

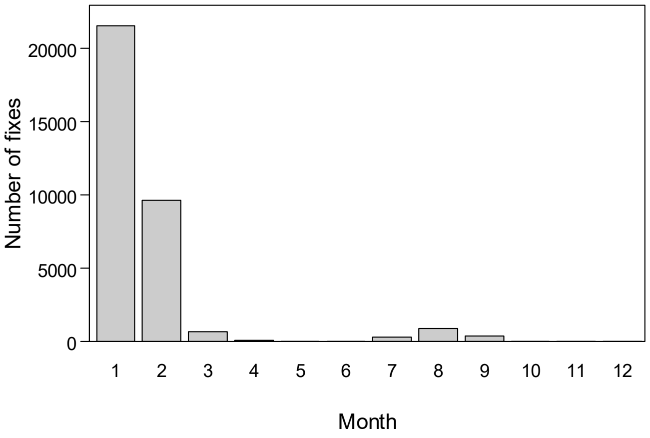


Figure 3: Monthly distribution of New Zealand sea lion fixes following grooming and filtering.

Two sea lions were tagged simultaneously with both an *Argos* satellite tag and a GPS tag. Comparison of the tracks from both tag types provides an opportunity to gauge the accuracy of the satellite tags relative to the highly accurate GPS tags. In general, the satellite tag tracks faithfully reproduced the 'real' GPS tag tracks (Figure A1-4). Close inspection reveals occasional deviations of the satellite tracks, but they did not materially affect the interpretation of track direction, geometry, or distance from the tagging site. In the rest of this report, satellite and GPS tag tracks were analysed together.

Sea lions tagged at Figure of Eight Island foraged in a distinctly different region from animals tagged at the other two sites: most fixes came from the southeast, south and southwest of the Auckland Islands shelf, and nearly all fixes were from within the 12 n. mile marine mammal sanctuary (Figure 4). However, we caution that only four Figure of Eight sea lions were tagged, and all were tagged during one season and year (summer 2008), so the results may not be representative. Sandy Bay and Dundas Island sea lions foraged widely over the northern Auckland Islands shelf and to the west of the Auckland Islands, often travelling beyond the 250 m isobath (Figure 4). Although the foraging ranges of Sandy Bay and Dundas Island sea lions overlapped strongly, the Dundas animals rarely visited the area west of the Auckland Islands, and they tended to focus on the eastern part of the northern Auckland Islands shelf.

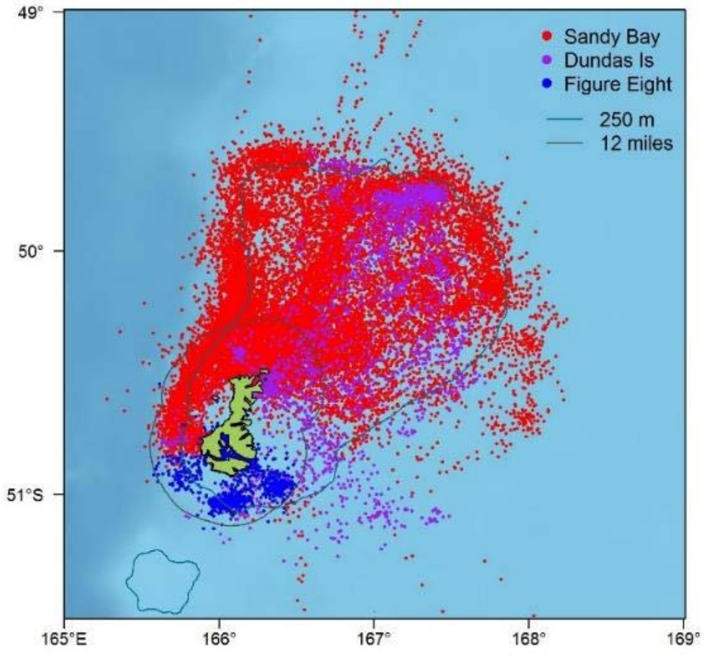


Figure 4: Groomed and filtered fix locations for sea lion tagged at three different colonies. Some fixes were beyond the map boundaries.

The SSSM with a 0.3-day time step produced a smoothed and regularised track that faithfully represented the raw data (Figure 5). Examination of individual tracks showed a similar close correspondence between the groomed filtered tracks, and the SSSM tracks, although the SSSM tracks occasionally overshot the raw data close to the Auckland Islands (Figure A1-5; e.g., see track 2007.1757.2). For sea lions that were tracked on several foraging trips within a season, each trip tended to follow a similar route, and the outward and return legs of each trip were also similar. However, some animals travelled different routes on different trips (Figure A1-5, track 2007.19147).

Most foraging trips remained on or near the Auckland Islands shelf (Figure 5 and Figure A1-5). However, five animals (3.5% of tracks) made long distance movements well beyond the shelf edge (Figure A1-6): two sea lions travelled north to the Stewart Island/Snares Islands shelf, one sea lion travelled to Otago Peninsula via Stewart Island (Figure A1-6), and two sea lions travelled southeast to the Campbell Island shelf. Two individuals that made long-distance journeys returned to the tagging site at Auckland Islands, both of which were lactating females at the time of tagging (Figure A1-6). The remaining three sea lions (two juvenile males and one adult male) did not return to Auckland Islands for as long as their tags were transmitting data.

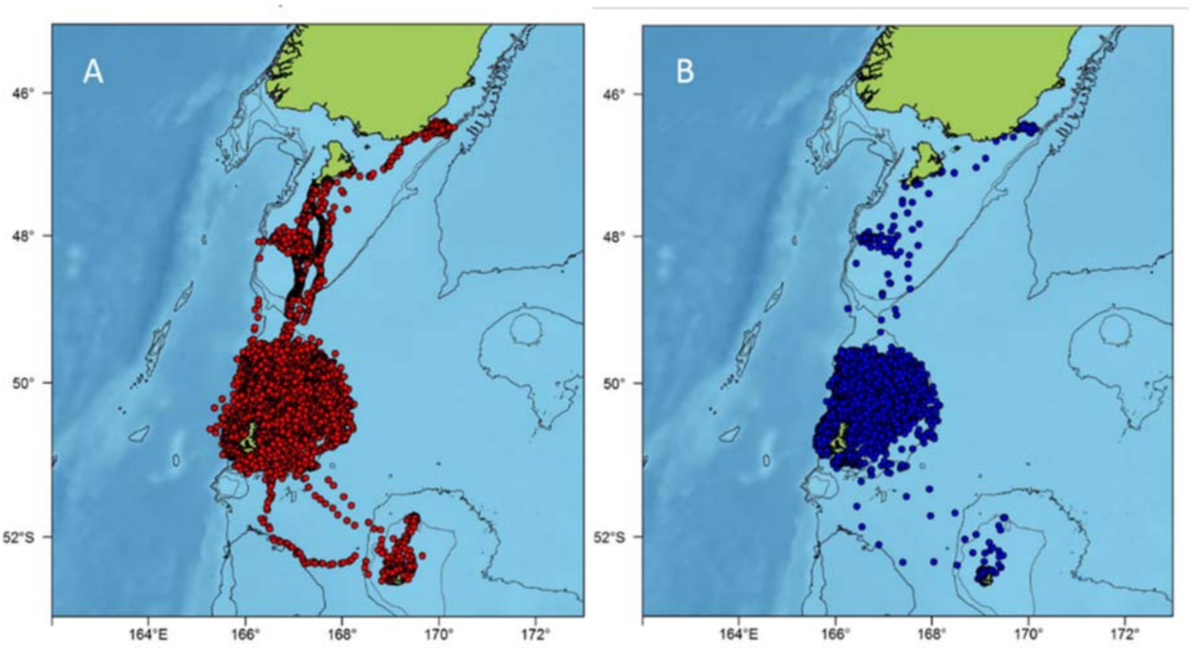


Figure 5: Comparison of A) groomed, filtered fixes and B) SSSM fitted locations (0.3-day time step) for all tagged sea lions combined.

3.2 Spatial density of female New Zealand sea lions

3.2.1 Model diagnostics

The quantile-quantile plots indicated that negative binomial models were appropriate for all demographic groups (Figure A2-2). The random effects diagnostic plots indicated that the assumption of a normally distributed sample size by individual was met for each demographic group (right-hand plots of Figure A2-3). All GAM smoother terms were significant ($p < 0.05$) for all models, except for lactating females at Figure of Eight Island. For this model, GAM smoothers for depth and distance to colony were not significant at this level, though were retained in the predictive model for this demographic group. Models for lactating females at Sandy Bay and Dundas Island explained 28% and 31% of the deviance, respectively, compared with 68% for the Figure of Eight Island model. The model for juvenile females explained 39% of the deviance (Table 8).

Table 8: Summary of models used to predict the spatial density of different demographic groups of female New Zealand sea lions at the Auckland Islands

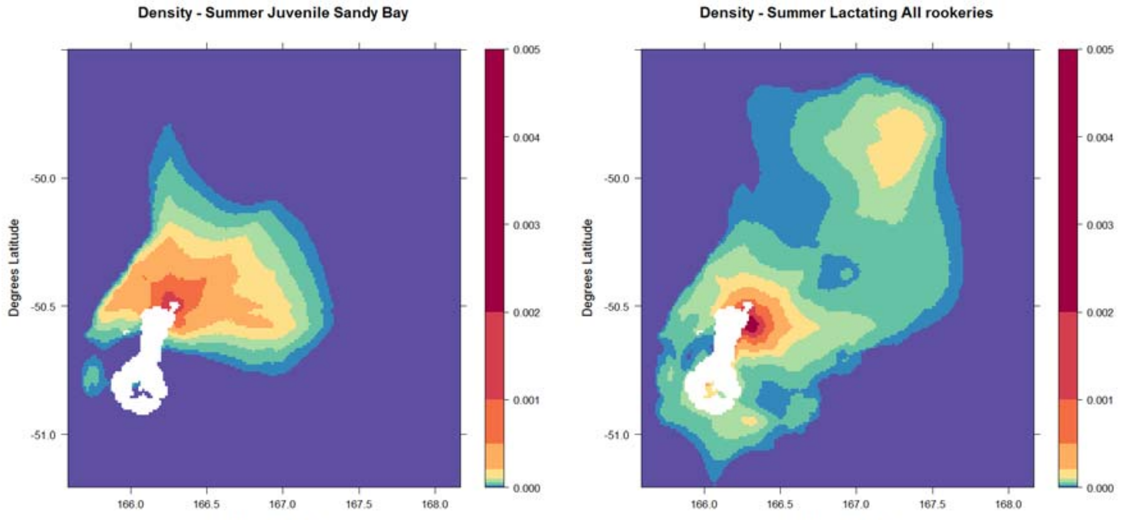
Modelled demographic group	Percentage of deviance explained by model
Juvenile females at Sandy Bay, Enderby Island	39.2
Lactating females at Sandy Bay, Enderby Island	27.7
Lactating females at Dundas Island	30.6
Lactating females at Figure of Eight Island	67.9

3.2.2 Model predictions

A visual inspection of the GAM splines shows that the density of New Zealand sea lion presences (per 1×1 km grid cell) decreased with increasing depth and distance from the rookery of origin (Figure A2-3). Note that the estimation of a bivariate spline for latitude and longitude will have been confounded with depth and distance to rookery effects to an unknown extent. This means that the actual relationship of New Zealand sea lion density with depth and distance to rookery may be different from those indicated by the model splines (Figure A2-3), though this should not adversely affect the quality of the prediction in horizontal space using these models, given that this will be accounted for by the bivariate spline for latitude and longitude.

The spatial predictions for each rookery are shown alongside spatial plots of the CVs associated with the respective predictions (Figure A2-4). Estimated spatial densities agreed well with the respective spatial density of satellite fixes for each demographic group (comparing Figure A2-4 with Figure 4). Relative to other demographic groups, the bulk of the estimated spatial density of lactating females at Figure of Eight Island and juveniles at Sandy Bay were close to the rookeries of origin. The CVs associated with spatial density prediction for lactating females at Sandy Bay were less than 0.25 at depths and regions where commercial trawl fisheries operate (comparing Figure A2-4 with Figure A3-1 to Figure A3-12). Larger CV values were obtained within fished areas for models for other demographic groups, which were fitted to smaller samples size of tracks (Table 6 and Table 7).

The combined estimated spatial density of female New Zealand sea lions extends across the entire Auckland Islands shelf and the upper reaches of surrounding slopes (Figure 6, bottom plot). Their foraging overlaps with commercial trawl fisheries in regions beyond the Auckland Islands Marine Mammal Sanctuary (compare Figure 6 with Figure A3-1 to Figure A3-12).



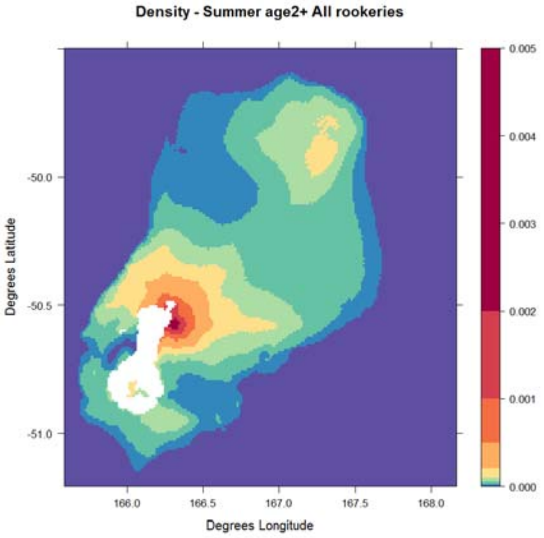


Figure 6: Estimated spatial density of female New Zealand sea lions of different demographic groups: juveniles (top-left), lactating females (top-right), and combined spatial density for all females age 2+ used in spatial risk models (bottom).

3.3 Spatial fisheries risk assessment

3.3.1 Commercial fishery effort and captures

Commercial fisheries protected species captures data were received on the 3rd July 2018 from Dragonfly Data Science, via Fisheries New Zealand. The number of observed New Zealand sea lion captures by fisheries group and fishing year from 1992/93 to 2016/17 is shown in Table A3-1 (SQU_{BT}), Table A3-2 (SQU_{MW}) and Table A3-3 (SCI and OTH). In the squid fishery (SQU_{BT} and SQU_{MW} combined), the number of observed trawl events exceeded 150 in all years since 1992/93 and exceeded 500 trawl events in all years but one since 2000/01. In all, 14 368 squid trawl events were observed in SQU 6T out of a total of 45 635 (observed and unobserved). A total of 235 New Zealand sea lion captures were reported by government fisheries observers across all observed squid fishing effort, of which 142 were reported to be females (note that a single unsexed capture was included in female captures for the purposes of risk modelling). A comparable percentage of female captures was obtained from observed bottom trawls (65%) and midwater trawls targeting squid (59%) (Table A3-1 and Table A3-2).

For scampi trawls, 2928 out of a total of 31 408 trawl event were observed since 1992/93, with two fishing years with no observer coverage (2004/05 and 2014/15). A total of 14 out of 16 captures (88%) on observed scampi trawl events were females. For trawls targeting all other species, 1113 out of 7242 trawl events were observed, with two observed captures (both female). There were no years without any observer coverage, though fewer than 50 trawl events were observed in 16 out of 25 years (Table A3-3).

The commercial fishery effort and female captures data are plotted spatially in Appendix 3, for SQU_{BT} (Figure A3-1 to Figure A3-3), SQU_{MW} (Figure A3-4 to Figure A3-6), SCI (Figure A3-7 to Figure A3-9) and OTH fishery groups (Figure A3-10 to Figure A3-12). These indicate that the spatial distribution of observer coverage approximately matched that of total effort for most years of the squid fishery. This was less often true for trawls targeting scampi and other species, which generally had a much lower observer coverage rate (Table A3-3).

3.3.2 Risk model diagnostics

SEFRA model diagnostics can be found in Appendix 4. MCMC mixing for all model parameters was good across all four MCMC chains (Figure A4-1). The prior and posterior distributions were equivalent for the population size (N_t) and capture event observability parameters (p_g^{obs}) (Figure A4-2). The posterior updates the prior for the catchability parameters (υ_g) and the probability of live capture parameter (ψ) (Figure A4-2). The model fits the mid-range of captures in each fishery reasonably well (Figures A4-3 and A4-4). For squid trawl fisheries, the model tends to over-predict captures at smaller values of observed captures and under-predict the larger values of observed captures (Figure A4-3). This trend is more evident in the midwater trawl squid fishery than the bottom trawl squid fishery. The same pattern in model fits was obtained for the scampi and other trawl fishery groups (Figure A4-4).

3.3.3 Risk model outputs

3.3.3.1 Catchability

Catchability is the probability of observable capture of New Zealand sea lions relating to a fishery group, given the spatial overlap of fisheries with the sea lions. The median and variance of catchability was highest for fisheries targeting species other than squid and scampi (OTH), followed by the scampi fishery group (SCI) (Figure 7). For squid fisheries, estimates followed a similar pattern for bottom and midwater trawls, i.e., similar catchability estimates for trawls not using SLEDs and those using non-standard SLEDs (all SLEDs prior to 2009), and then much lower catchability estimates for trawls using standardised SLEDs (all since 2009) (Figure 7). All catchability parameter estimates for the base case model run are presented in Table A5-1.

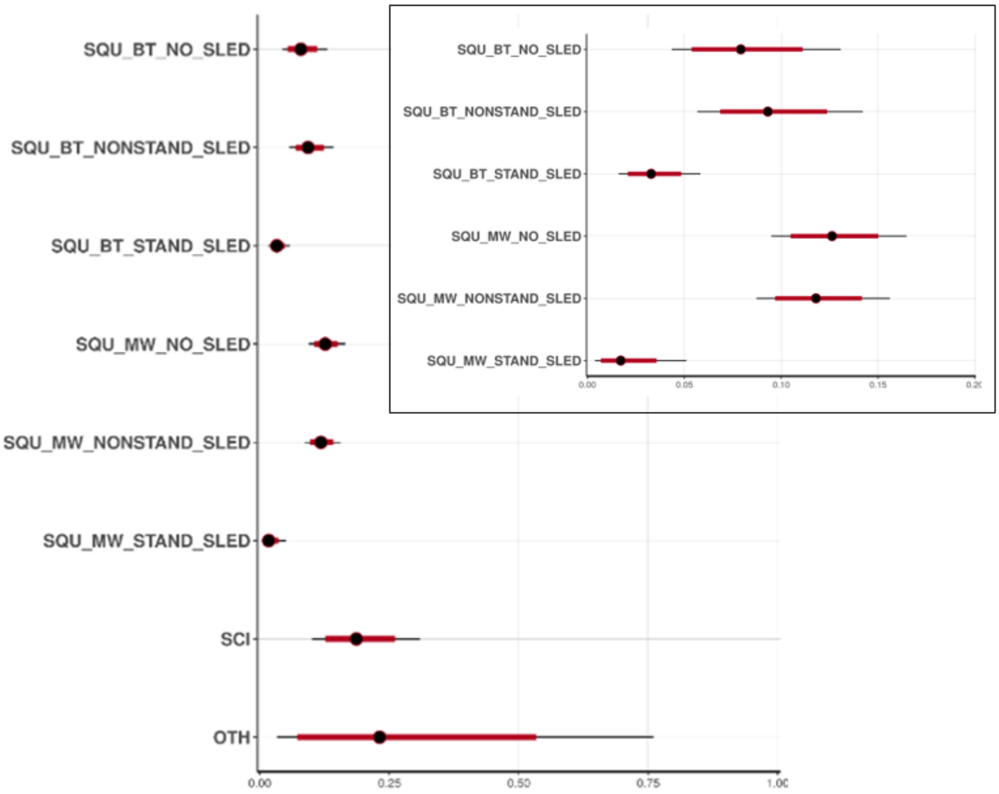


Figure 7: Posteriors of risk model estimates of catchability of female New Zealand sea lions to Auckland Islands commercial fishery groups: "SQU_BT_NO_SLED" = bottom trawls targeting southern arrow squid without a sea lion exclusion device (SLED), "SQU_BT_NONSTAND_SLED" = bottom trawls targeting squid with a SLED using a non-standardised configuration, "SQU_BT_STAND_SLED" = bottom trawls targeting squid with a SLED using a standardised configuration, other groups containing "MW" instead of "BT", were as above except that a midwater trawl was used, "SCI" = scampi trawl, "OTH" = trawls targeting all other species at the Auckland Islands. Posteriors for fishery groups targeting southern arrow squid are also shown in an embedded plot, with the x-axis rescales to make outputs easier to read.

Cryptic mortality multipliers relating to the use of standardised SLEDs in the SQU_{BT} and SQU_{MW} fisheries were estimated by Meyer (2019). These posterior distributions were, in part, informed by the ratio of catchability estimates for fishery groups using standardised SLEDs ($SQU_{BT\_STAND\_SLED}$ and $SQU_{MW\_STAND\_SLED}$) to those without SLEDs ($SQU_{BT\_NO\_SLED}$ and $SQU_{MW\_NO\_SLED}$), which Meyer (2019) used to simulate the exit probability of New Zealand sea lions passing through a SLED. This ratio was lower for midwater trawls than for bottom trawls, indicating that the SLED exit probability was higher for midwater trawls (Figure 8). Note that the higher estimated SLED exit rate in midwater trawls is somewhat offset by a slightly higher estimated interaction rate (i.e., captures per unit overlap in trawls without SLEDs, Figure 7) and a higher estimated cryptic mortality rate (Meyer 2019). For these reasons, estimated deaths per unit overlap are comparable for both methods.

Prior distributions for the probability that a capture event was observable by fishery group (p_g^{obs}) were then derived from the reciprocal of the generated *cryptic multiplier* posterior samples for bottom trawls and midwater trawls from Meyer (2019). These values of p_g^{obs} were then used to estimate informed beta priors for this parameter (Table 5).

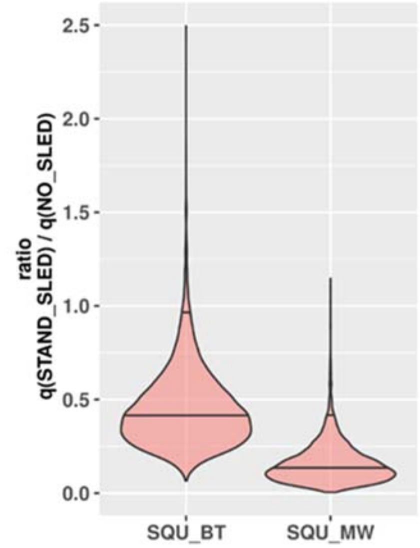


Figure 8: Estimated ratio of catchabilities estimates for squid trawls using standardised sea lion exclusion devices (SLEDs) compared with squid trawls without SLEDs. Separate estimates were obtained for bottom trawls (BT, left) and midwater trawls (MW, right). These ratios are not used by the spatial risk model, though they informed capture event observability prior distributions estimated by Meyer (2019).

3.3.3.2 Estimated 'current' annual deaths

The convention from pervious implementations of the SEFRA approach has been to use quantities obtained for the final three model years to estimate 'current' annual deaths and risk ratios (e.g. Abraham et al. 2017; Roberts et al. 2019). SEFRA estimates of current female annual deaths (i.e., from 2014/15 to 2016/17) were slightly higher for the scampi fishery (3.00 individuals yr^{-1}, 95% credible interval or CI = 1.00–6.33) than for the squid fishery (2.33 individuals yr^{-1}, 95% CI = 0.67–5.00). Estimated annual deaths were comparatively low for trawls targeting all other species (0.33 individuals yr^{-1}, 95% CI = 0.00–1.33). The estimate of annual deaths across all trawl fisheries was 6.00 (95% CI = 3.00–10.00) (Table **9** and Figure 9).

Table 9 : Risk model estimates of annual deaths for commercial trawl fishery groups, showing the median, 90% and 95% credible intervals. These estimates were produced by the base case risk model using $\phi = 0.10$. Estimates of annual deaths were insensitive to assuming alternative values of ϕ (not shown here).

	Quantiles of estimated annual deaths 2014/15 to 2016/17				
Fishery group	**2.5%**	**5.0%**	**50.0%**	**95.0%**	**97.5%**
All trawl fisheries	3.00	3.33	**6.00**	9.33	10.00
SQU	0.67	1.00	**2.33**	4.67	5.00
SCI	1.00	1.33	**3.00**	5.67	6.33
OTH	0.00	0.00	**0.33**	1.00	1.33

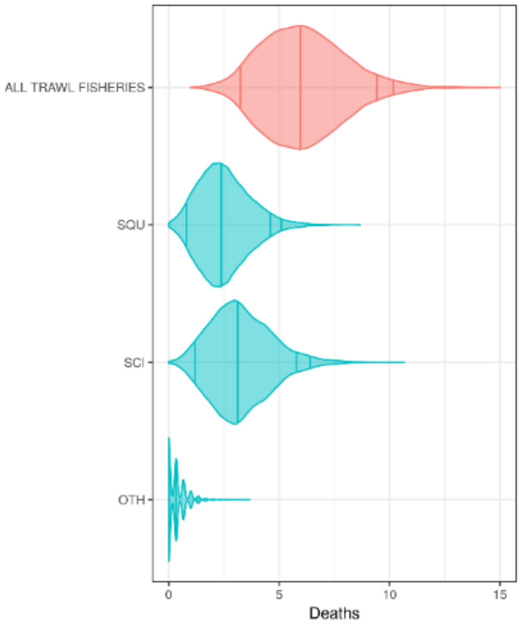


Figure 9: Estimated annual deaths of female New Zealand sea lions in Auckland Islands commercial trawl fisheries: all trawl fisheries, “SQU” = southern arrow squid trawl (SQU 6T), “SCI” = scampi trawl (SCI 6A), and “OTH” = trawls targeting all other species. The median, 2.50%, 95.0%, and 97.5% quantiles are indicated as vertical lines within each density.

3.3.3.3 Estimated ‘current’ annual population risk

For all commercial trawl fisheries combined, the ‘current’ (from 2014/15 to 2016/17) risk ratio (calculated as estimated deaths as a proportion of the estimated PST) was 0.24 (95% CI = 0.11–0.47) when applying a default calibration coefficient (ϕ) value of 0.10, corresponding to a population outcome at 95% of un-impacted status (Table 10 and Figure 10). That is, for all commercial trawl fisheries combined, the best estimate of annual mortalities for the assessed commercial fisheries did not exceed the annual PST between 2014/15 and 2016/17, indicating that the recent mortality levels for these fisheries would not depress the equilibrium population below 95% of unimpacted status. For each of the fishery groups the upper 95% credible interval of the risk ratio, assuming a calibration coefficient (ϕ) value of 0.10, did not exceed 0.26, i.e., 0.21 for the squid trawl fishery, 0.26 for the scampi trawl fishery, and 0.06 for other trawl fisheries.

Sensitivity model runs using alternative calibration coefficient (ϕ) values of 0.05 and 0.20 (consistent with population recovery to at least 97.5% and 90% of unimpacted status, respectively) produced risk ratio estimates approximately double or half (respectively) of those obtained from the base run (Table **10** and Figure 9). The upper 95% credible interval of the risk ratio was below 1 for these sensitivity runs, with a maximum of 0.51 for a single fishery group (scampi trawls) (Table **10**).

Table 10 : Risk model estimates of annual risk ratio for commercial fishery groups, assuming the base case value of ϕ (0.10), and sensitivity values of ϕ (0.05 and 0.20).

PST calibration coefficient (ϕ)	Fishery group	Quantiles of estimated risk ratio 2014/15 to 2016/17				
		2.5%	**5.0%**	**50.0%**	**95.0%**	**97.5%**
0.10	All trawl fisheries	0.11	0.13	**0.24**	0.38	0.47
0.10	SQU	0.02	0.03	**0.09**	0.19	0.21
0.10	SCI	0.04	0.05	**0.13**	0.23	0.26
0.10	OTH	0.00	0.00	**0.01**	0.05	0.06
0.05	All trawl fisheries	0.22	0,25	**0.47**	0.77	0.84
0.05	SQU	0.05	0.07	**0.19**	0.37	0.41
0.05	SCI	0.08	0.10	**0.25**	0.46	0.51
0.05	OTH	0.00	0.00	**0.04**	0.09	0.12
0.20	All trawl fisheries	0.06	0.06	**0.12**	0.19	0.21
0.20	SQU	0.01	0.02	**0.05**	0.09	0.11
0.20	SCI	0.02	0.03	**0.06**	0.12	0.13
0.20	OTH	0.00	0.00	**0.01**	0.02	0.03

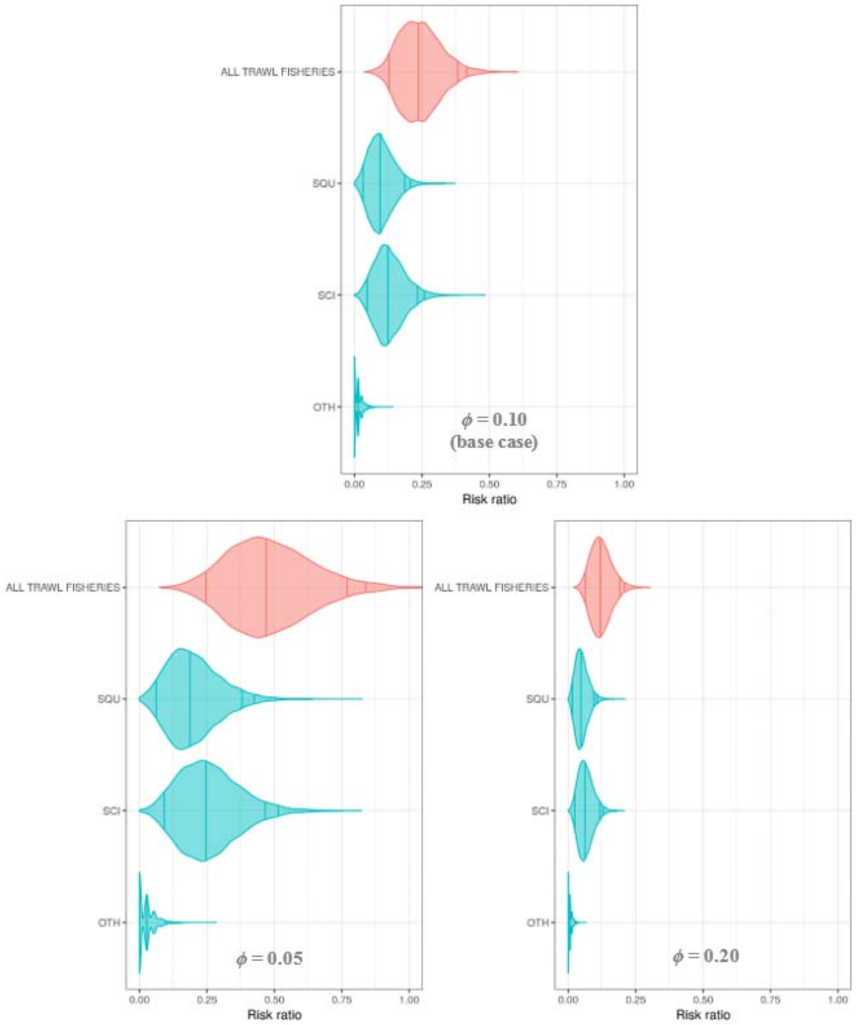


Figure 10: Annual commercial trawl fishery risk ratios for female New Zealand sea lions at the Auckland Islands. "SQU" = southern arrow squid trawl (SQU 6T), "SCI" = scampi trawl (SCI 6A), and "OTH" = trawls targeting all other species. Estimates from the base case risk model, assuming ϕ = 0.10 (top), and sensitivity values of ϕ = 0.05 (bottom-left) and ϕ = 0.20 (bottom-right). The median, 95.0% and 97.5% quantiles are indicated as vertical lines within each density.

3.3.3.4 Changes in deaths and risk through time

Trawls targeting southern arrow squid were estimated to have killed more female New Zealand sea lions that other Auckland Islands fisheries across the entire assessed time period (1993/93 to 2016/17) (Figure 11). The sum of the mean estimate of deaths across this period was 578 females in trawls targeting squid, 150 females in trawls targeting scampi, and 19 females in trawls targeting all other species (posteriors for fishery group-disaggregated deaths are shown in Figures 12 to 15). Prior to 2009, estimated annual deaths in the squid fishery were most responsive to changes in effort in the fishery (compare Figure 11 with Figures 12 and 13). The use of standardised SLEDs in the squid fishery affected a major reduction in estimated deaths since 2009, which were of a similar magnitude to estimated deaths in the scampi fishery across this time period (Figure 11). In addition, some of the reduction in estimated deaths through the assessed period will have resulted from the decline in female population size at the Auckland Islands (also displayed in Figures 12 to 15).

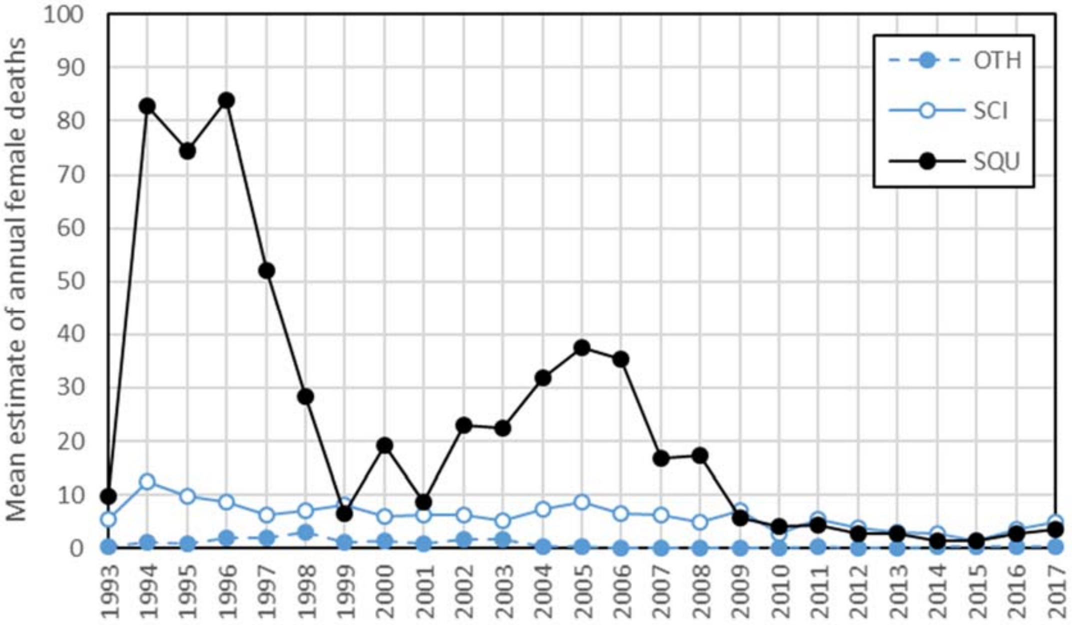


Figure 11: Mean estimate of annual deaths of female New Zealand sea lions in Auckland Islands trawl fisheries from 1992/93 to 2016/17, aggregated by target species: “SQU” = southern arrow squid, “SCI” = scampi, “OTH” = all trawls targeting all other species. This model run assumed a calibration coefficient (ϕ) of 0.1.

The total number of fishing events, spatial overlap, female New Zealand sea lion annual deaths, and annual risk ratio were estimated for each fishery, for individual years from 1992/93 to 2016/17 (Figures 12 to 15). The median risk ratio was below 1 for all years in each of the four trawl fishery groups (SQU_{BT}, SQU_{MW}, SCI and OTH) except for SQU_{MW} from 1993/94 to 1995/96, when the median risk ratio was close to 1.5 (Figure 13). Overlap per unit effort did not change much over time for either bottom or midwater squid trawls (SQU_{BT} and SQU_{MW}). Effort was higher for midwater squid trawls in the mid-1990s with approximately 3000 trawl events annually, declining to less than 500 annually since 2010. For bottom trawls targeting squid, effort was at or below 1000 trawl events annually since 2010, but with peaks of approximately 1500 annually in the mid-1990s and the mid-2000s. Increases in annual deaths and risk ratio were consistent with increased effort, particularly when this was combined with increased overlap per unit effort and a higher population size. These trends were most evident in the period prior to the introduction of standard SLED use, i.e., prior to 2009.

The median of annual risk ratio estimates remained below 0.5 in all assessed years of the scampi fishery (Figure 14). Spatial overlap per unit effort in this fishery has remained consistent over time. Changes in the risk ratio, particularly in the last ten years, when the annual female population size has remained stable, are small and coincide with decreasing then increasing trends in fishing effort. For trawls targeting all other species, the annual risk ratio has been below 0.1 since 1992/93 and close to zero since 2003/04 (Figure 15). Annual effort (at or less than 200 trawl events annually) is much less in these other fisheries compared to the squid and scampi trawl fisheries.

4. DISCUSSION

4.1 Model inputs

The foraging of female New Zealand sea lions extends across the entire Auckland Islands shelf and upper slope regions, and overlaps with commercial trawl fisheries in regions beyond the Auckland Islands Marine Mammal Sanctuary (Figure 6). Based on the available sample of tracking data, the at-sea foraging patterns of females at Sandy bay, Dundas Island, and Figure of Eight Island are likely to be quite different. This corroborates the findings of previous analyses using some of the same data (Chilvers 2009; Chilvers et al. 2011). However, relative to their contribution to the total Auckland Islands population, a disproportionate amount of the existing New Zealand sea lion tracking data was obtained from juvenile and lactating females tagged at Sandy Bay, Enderby Island (Table 6 and Table 7). Non-breeding females at mature ages have not previously been a focal demographic for field-based tracking studies at the Auckland Islands, and their foraging patterns (and overlap with fishing) have not been estimated to date.

In addition, there has been no tracking of juveniles at Dundas Island, which will comprise a relatively large proportion of the population size at ages vulnerable to capture in trawls (DOC 2019). The spatial density estimate obtained for juvenile females at Sandy Bay indicates that they remain closer to their rookery of origin than lactating females (comparing the top two plots of Figure 6), such that the degree of spatial overlap with commercial fisheries may be quite different for Dundas Island juveniles. In addition, the number of individuals tagged in winter months is low for producing a representative estimate of spatial density for this period (Figure 3), which could be used to develop a seasonal SEFRA model. Previous research based on a small sample of tracks has indicated that the dive profiles may be quite different for winter foraging (Chilvers et al. 2013). Future monitoring of New Zealand sea lion foraging could be targeted to plug these information gaps.

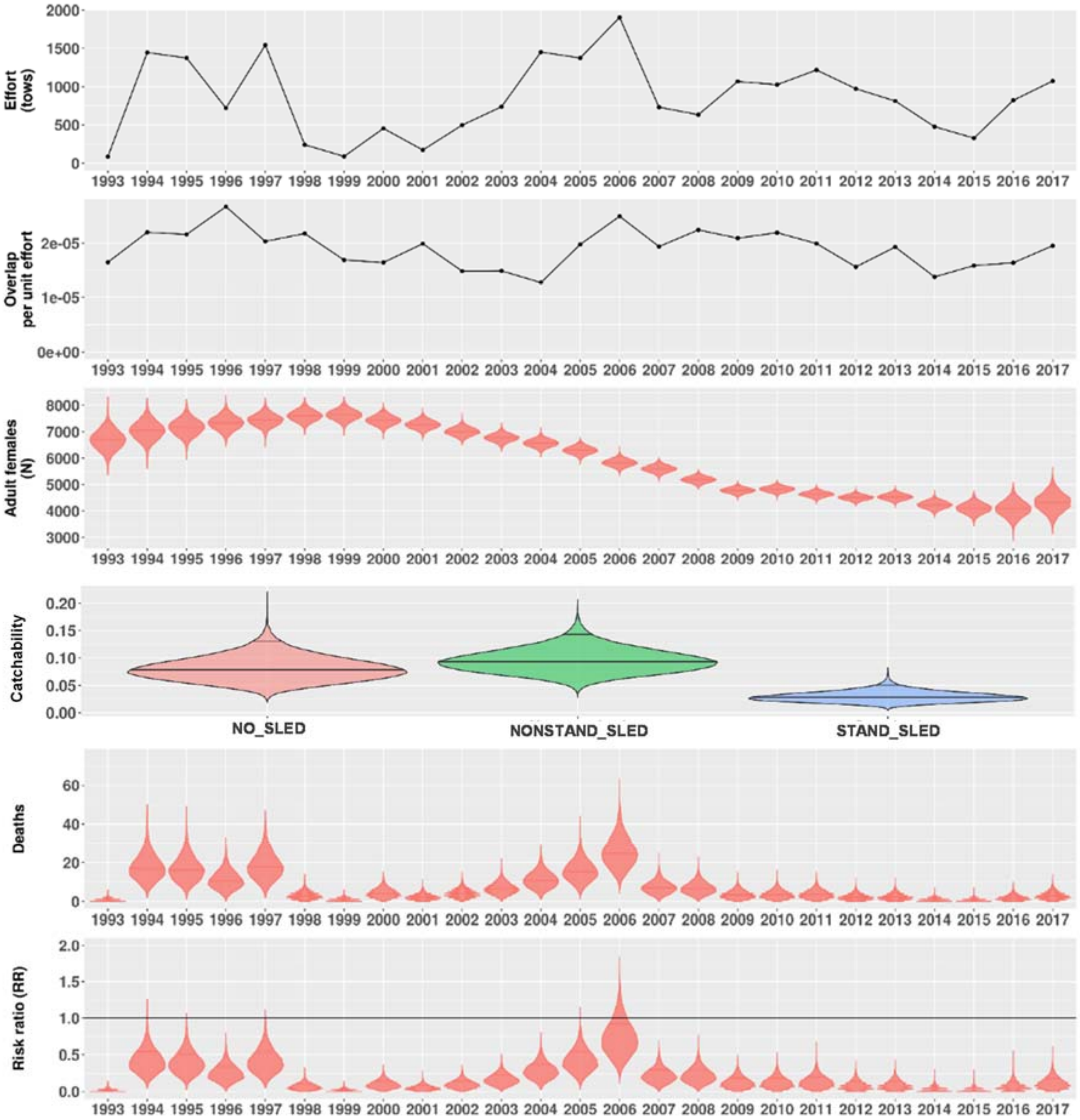


Figure 12: Effort of bottom trawls targeting southern arrow squid (SQU_{BT}) by fishing year (top) and spatial risk model outputs for female New Zealand sea lions for this fishery: spatial overlap per unit effort, female population size, catchability ("NO_SLED" = trawls with no sea lion exclusion device—all effort prior to 2001 and some effort from 2001 to 2008; "NONSTAND_SLED" = trawls with non-standardised SLED—most trawls from 2001 to 2008; "STAND_SLED" = trawls with a standardised SLED—all trawls since 2009), annual deaths and risk ratio. This model run assumed a calibration coefficient (ϕ) of 0.1.

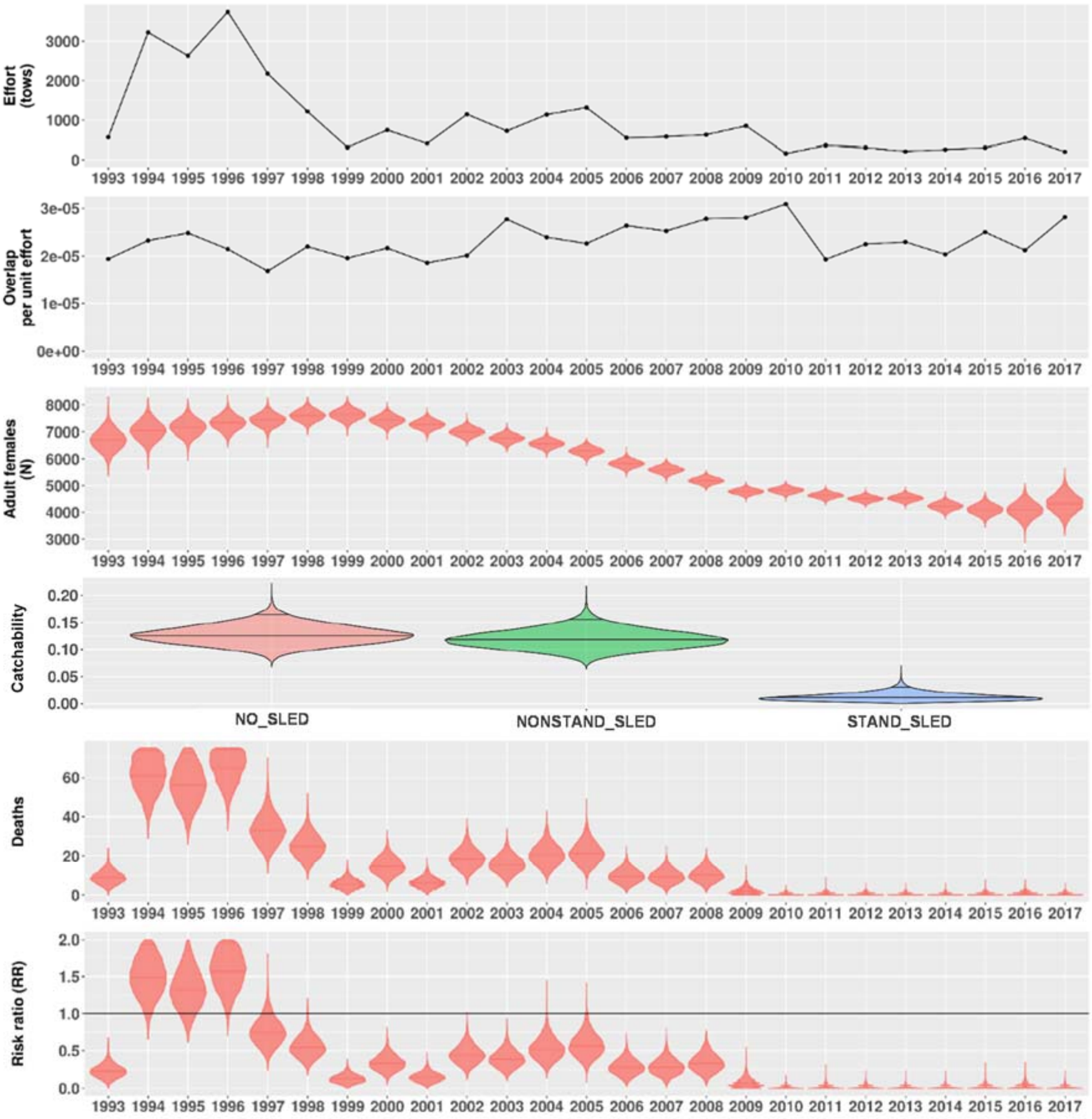


Figure 13: Effort of midwater trawls targeting southern arrow squid (SQU_{MW}) by fishing year (top) and spatial risk model outputs for female New Zealand sea lions for this fishery: spatial overlap per unit effort, female population size, catchability (“NO_SLED” = trawls with no sea lion exclusion device—all effort prior to 2001 and some effort from 2001 to 2008; “NONSTAND_SLED” = trawls with non-standardised SLED—most trawls from 2001 to 2008; “STAND_SLED” = trawls with a standardised SLED—all trawls since 2009), annual deaths and risk ratio. This model run assumed a calibration coefficient (ϕ) of 0.1. Risk ratios higher than two were not displayed on this plot to make risk scores easier to read across all years.

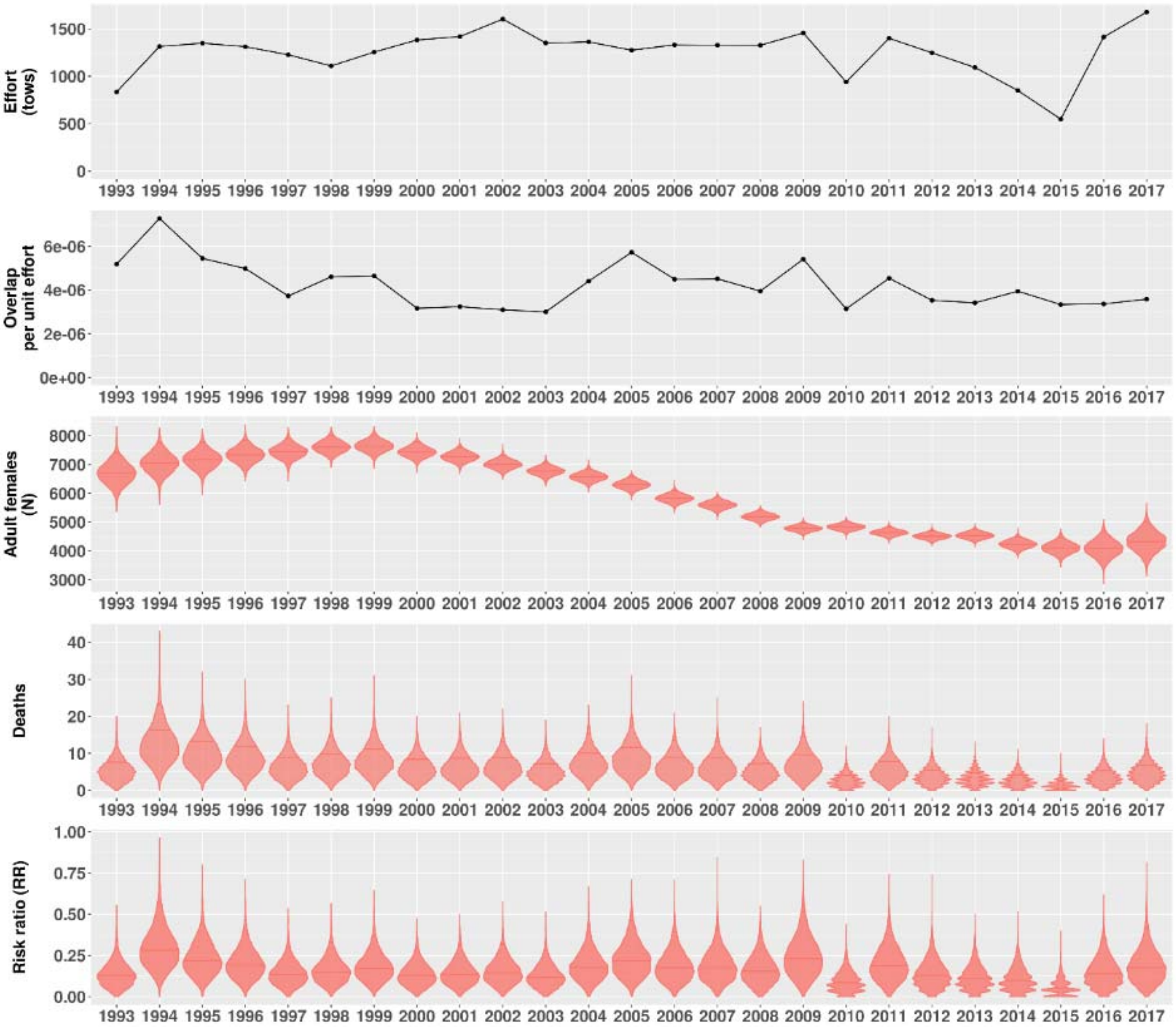


Figure 14: Effort of trawls targeting scampi (SCI) by fishing year (top) and spatial risk model outputs for female New Zealand sea lions for this fishery group also by fishing year: spatial overlap per unit effort, female population size, annual deaths and risk ratio. This model run assumed a calibration coefficient (ϕ) of 0.1.

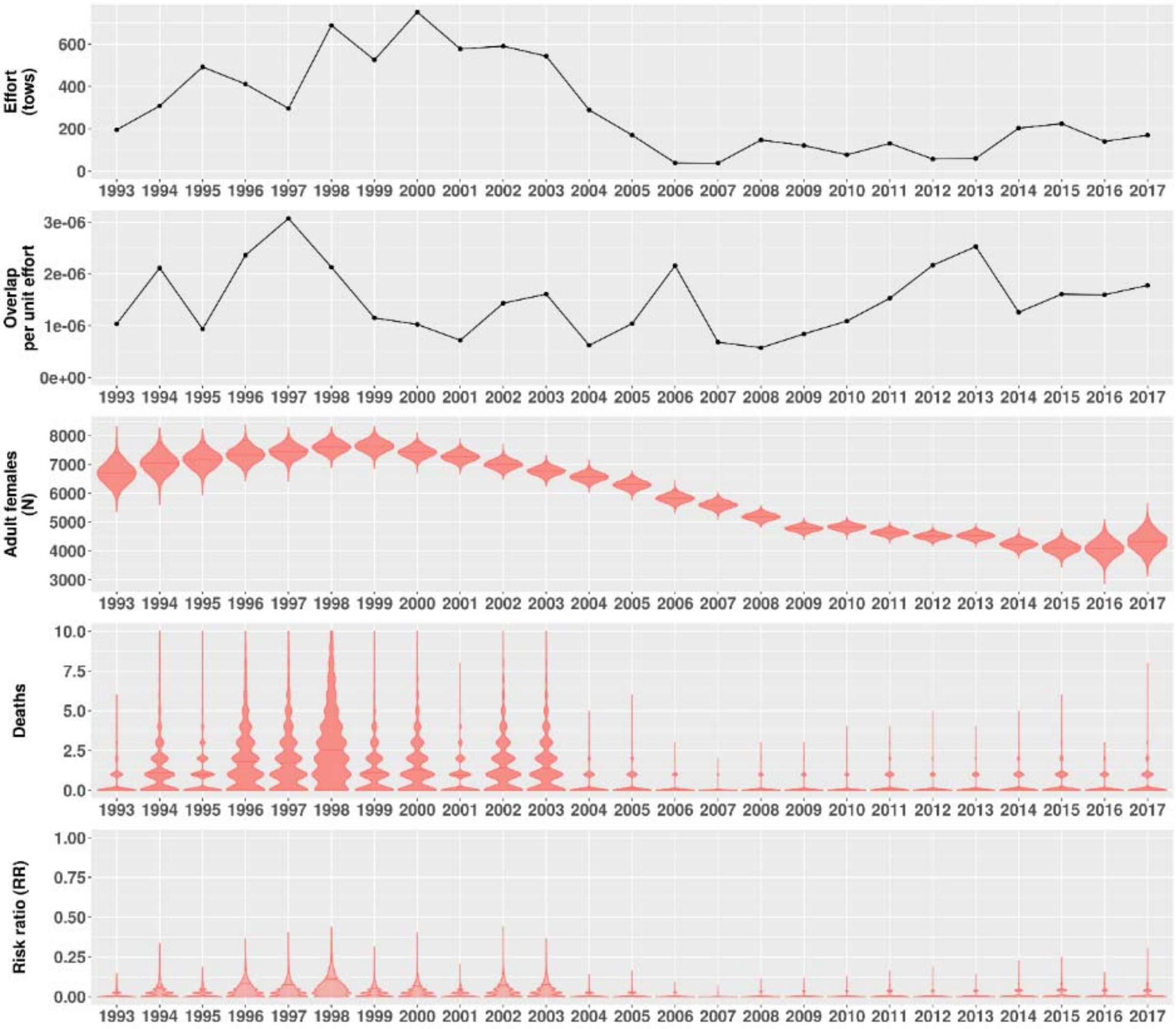


Figure 15: Effort of trawls targeting all other species (OTH) by fishing year (top) and spatial risk model outputs for female New Zealand sea lions for this fishery group also by fishing year: spatial overlap per unit effort, female population size, annual deaths and risk ratio. This model run assumed a calibration coefficient (ϕ) of 0.1.

The habitat models developed by this assessment were relatively simple. Their reliance on latitude and longitude predictors means that they have no predictive power outside of the Auckland Islands area. Initial explorations with the development of habitat models using only true habitat variables (Mark Hindell, unpublished data) were constrained by a lack of prey density information in key foraging areas, e.g., along the western shelf break of the Auckland Islands. The seafloor in this region is likely to be too rough for commercial or survey bottom trawling (Roberts et al. 2018), although midwater trawls and acoustic methods could be used to estimate the relative prey densities in these areas. These data could then be used to develop habitat models with improved predictive power, i.e., that could be used to estimate spatial densities at other locations, or to estimate the effects of climate-mediated changes in prey distribution on the foraging patterns of New Zealand sea lions, and their overlap with fisheries.

Time Depth Recorders (TDRs) have also been deployed on female New Zealand sea lions at the Auckland Islands, simultaneously with satellite tags, which could potentially be used to produce separate spatial densities of midwater and demersal foraging (Meynier et al. 2014). However, the metadata required to reliably pair satellite tag and TDR data were lacking. If these metadata cannot be acquired, then future monitoring could repeat paired satellite tag/TDR monitoring with the objective of producing separate estimates of spatial overlap and risk for different dive types, which are likely to have differential catchability in midwater versus bottom trawls.

Demographic inputs included total female population size by year, which were obtained from population modelling by Roberts (2017) and were relatively precise (Table 4). In the absence of an estimate of r^{max} for New Zealand sea lions, the default value of 0.12 used by US stock assessments (Wade 1998) was assumed, with prior parameters arbitrarily specified to generate r^{max} values ranging from 0.10 to 0.14. SEFRA risk ratio estimates scale directly with the input value of r^{max}, such that estimates of risk will be highly sensitive to the value assumed. Species-specific estimates of r^{max} could be obtained using allometric invariants, e.g., following the approach of Dillingham et al. (2016), as implemented by Edwards et al. (2018) for estimating r^{max} for Hector's and Māui dolphins.

4.2 Spatial risk assessment

The SEFRA model propagates uncertainty with respect to the estimation of all model parameters though to annual estimates of overlap, deaths, and risk. However, this implementation of the SEFRA approach (and all previous implementations, e.g. Abraham et al. 2017; Roberts et al. 2019) does not propagate uncertainty with respect to the prediction of the spatial density of the study species. Future SEFRA models could consider using geostatistical modelling approaches that fit to spatial distribution information simultaneously with other data inputs (e.g., Thorson et al. 2015).

One advantage of the SEFRA approach is that it accounts for potential differences in the spatial overlap of the study species with total fishing effort versus the observed portion, which may otherwise bias the estimation of total captures. This is likely to be most beneficial where the level of observer coverage is low, as is the case for the Auckland Islands scampi fishery, or where the distribution of observer coverage is not spatially representative of total effort. For the squid trawl fishery, the number of observed tows is very high (Table A3-1 and Table A3-2), and SEFRA estimates of annual deaths are unlikely to be very different from those of spatially-blind assessments.

Another advantage of the SEFRA approach is that estimated overlap, deaths, and risk can be presented spatially at a user-defined resolution. This could then be used to assess the agreement between the estimated versus observed spatial distribution of captures (e.g., Roberts et al. 2019), which may point to aspects of the SEFRA model that could be improved. The identification of locations with relatively high risk ratios can also be used to guide conservation management.

The SEFRA model also provides a framework for estimating fishery-related deaths and risk in years lacking any observer coverage, e.g., the scampi fishery in 2004/05 and 2014/15, or years prior to 1992/93 (when observer coverage began for Auckland Islands fisheries). This assumes that the catchability of New Zealand sea lions is equivalent to periods with observer coverage, and the validity of this assumption may need consideration if extrapolating back or forward through time.

4.3 Female only risk assessment

A female-only SEFRA model was developed on the basis that New Zealand sea lions are polygamous breeders (Cawthorn et al. 1985) and that populations will be much more responsive to the anthropogenic deaths of females than males. It was found that captures in all trawl fisheries appear to be biased towards females, and this was particularly the case for the scampi fishery (Figure A3-1 to Figure A3-3). As such, it is recommended that future assessments follow the approach of estimating deaths and risk for female New Zealand sea lions, separately to that of males.

4.4 Changes in risk through time

Under the default assumption of linear density dependence, the equilibrium population outcome scales directly with the combined risk ratio (corresponding to the proportion of $r^{\max}$ that is killed). This allows the translation of risk scores other than 1 to a corresponding population outcome, i.e., using $\phi = 0.10$

in the calculation of the PST, a combined anthropogenic risk score of 1 corresponds to a population outcome at 95% of unimpacted status; a risk ratio of 0.5 corresponds to a population outcome at 97.5% of unimpacted status; and a risk ratio of 0.1 corresponds to a population outcome at 99.5% of unimpacted status.

When using the base case PST ($\phi = 0.10$), the annual risk ratio across all trawl fishery groups was 0.24 (95% CI = 0.11–0.47) for the period from 2014/15 to 2016/17. In the absence of other anthropogenic threats, future deaths at the upper 95% CI risk ratio would be consistent with population recovery to approximately 97.5% of unimpacted population size, when assuming logistic population growth. However, the median risk ratio estimate was close to two across all Auckland Islands fisheries from 1993/94 to 1995/96 (the upper 95% CI of the risk ratio estimate was close to three), indicating that fishery deaths may have had a greater impact on the New Zealand sea lion in the past (Table A6-19).

This is consistent with the outputs of demographic population modelling by Roberts & Doonan (2017) and Roberts (2019), which found that estimates of commercial fishery deaths were insufficient to drive the 40% decline in breeder numbers at the Auckland Islands from the late 1990s to 2009 (DOC 2019). However, note that the SEFRA assessment did not include fishing effort prior to 1992/93, which includes a period of relatively high squid fishery effort through most of the 1980s (Richard Wells, unpublished data). This SEFRA model could potentially be extended to include years prior to 1992/93 and estimate annual deaths and risk ratio from the start of fishing records at the Auckland Islands, in the late-1970s.

With respect to the Auckland Islands squid trawl fishery, there was a shift in the fleet composition from predominantly midwater trawls at the beginning of the period, to predominantly bottom trawls by the end. This shift was caused by the gradual replacement of midwater trawlers after the breakup of the Soviet Union in 1991 (Richard Wells, pers. communication). Any gear-specific effects on catchability and cryptic mortality were accounted for by this assessment, which estimated/specified these separately for midwater and bottom trawls (Figure 7 and Figure 8).

The existing information suggests that that the catchability of New Zealand sea lions in commercial trawls may vary through time (e.g., the sudden shift in the capture rate of New Zealand sea lions in Campbell Island trawl southern blue whiting (*Micromesistius australis*) fishery). This variation could occur as a result of changes in sea lion behaviour, such as in response to nutritional status-mediated changes in sea lion behaviour around fishing vessels, or as a result of potentially unaccounted-for changes in fishing fleet composition or operations. Continued observer coverage can be used to assess potential changes in catchability through time, or to provide the information requirements for developing potential covariates of catchability, e.g., to account for potential changes in tow duration through time. Changes in the spatial foraging patterns of New Zealand sea lions may occur in response to changes in prey distribution, affecting the degree of overlap with fishing operations. On this basis, it would be advantageous to collect information that can be used to assess potential changes in spatial foraging patterns of New Zealand sea lions at the Auckland Islands in future years.

5. CONCLUSIONS AND FUTURE RESEARCH

The primary conclusions of this research are as follows:

- The estimated spatial density of female New Zealand sea lions varies with demographic group (age and rookery of origin) and extends across the entire Auckland Islands shelf and the upper reaches of surrounding slopes. Their foraging overlaps with commercial trawl fisheries in regions beyond the Auckland Islands Marine Mammal Sanctuary.

- The SEFRA model estimate of annual trawl fishery deaths across all fishery groups (targeting squid, scampi and all other species) was 6 females (95% CI = 3–10) for the period 2014/15 to 2016/18. When using the base case PST (ϕ = 0.10), the annual risk ratio for this period was 0.24 (95% CI = 0.11–0.47). In the absence of other anthropogenic threats, future deaths at the upper 95% CI risk ratio would be consistent with population recovery to 97.5% of unimpacted population size, when assuming logistic population growth.

- Squid fishery deaths dominated the deaths of female New Zealand sea lions in Auckland Islands trawl fisheries throughout the period that SLEDs were not used (mostly prior to 2001) or were used but were not standardised (2001 to 2008). The ubiquitous use of standardised SLEDs since 2009 affected a major reduction in annual deaths, which, since then, have been similar in magnitude to estimated deaths in the scampi fishery.

- The median risk ratio was close to two across all Auckland Islands fisheries from 1993/94 to 1995/96 (upper 95% CI close to 3, indicating that fishery deaths may have had a greater impact on the New Zealand sea lion in the past, although prior to the 40% decline in breeder numbers since the late 1990s. The model did not include the 1980s period when effort in the squid trawl fishery was consistently higher than in the period addressed by this assessment.

Potential future research addressing points in the above discussion and conclusions are as follows:

- In order to provide the information requirements of a seasonal model (which will be most influential for the assessment of scampi fishery risk), monitor the at-sea foraging of female New Zealand sea lions at Dundas Island in winter months. Also monitor the foraging of juveniles at Dundas Island, which has not previously been assessed.

- The SEFRA model could be extended to estimate annual deaths and risk using geolocated fishing effort data for fishing years without observer coverage prior to 1992/93. This assessment would ideally consider whether the catchability of sea lions in trawls without SLEDs is likely to be the same as that estimated from the observed period.

- The SEFRA model outputs can be used to produce spatial plots of overlap (between fishing and New Zealand sea lions), deaths, and risk, which can be used to validate model outputs and inform conservation management.

- Consider the development of geostatistical risk models, that simultaneously predict the spatial density of New Zealand sea lions (and other protected species) along with the estimation of model parameters during risk model optimisation. This would allow for the propagation of uncertainty associated with the estimation of spatial density through to the estimation of annual overlap, deaths, and risk.

- More comprehensive information on the spatial information of key prey species (e.g., areas where female New Zealand sea lions forage though prey distribution information is totally lacking) may be required to inform the development of habitat models with high predictive power that could potentially be applied to other populations.

6. ACKNOWLEDGMENTS

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APPENDIX 1 ADDITIONAL PLOTS FROM PROCESSING OF NEW ZEALAND SEA LION TRACKING DATA

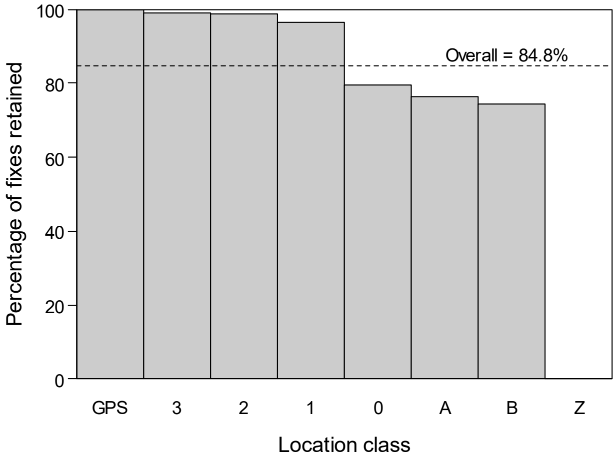

Figure A1-1: Percentage of fixes retained in each location class after speed filtering. The dashed line indicates the overall percentage for all classes combined.

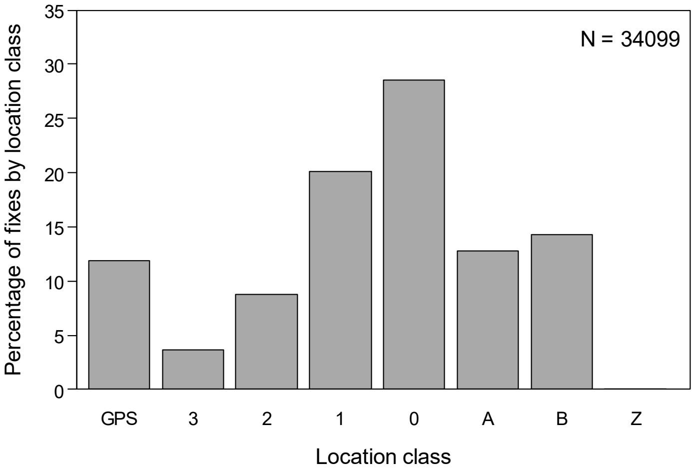

Figure A1-2: Percentage of sea lion location fixes in each location class following grooming and filtering (see Table 1 for definition of location classes).

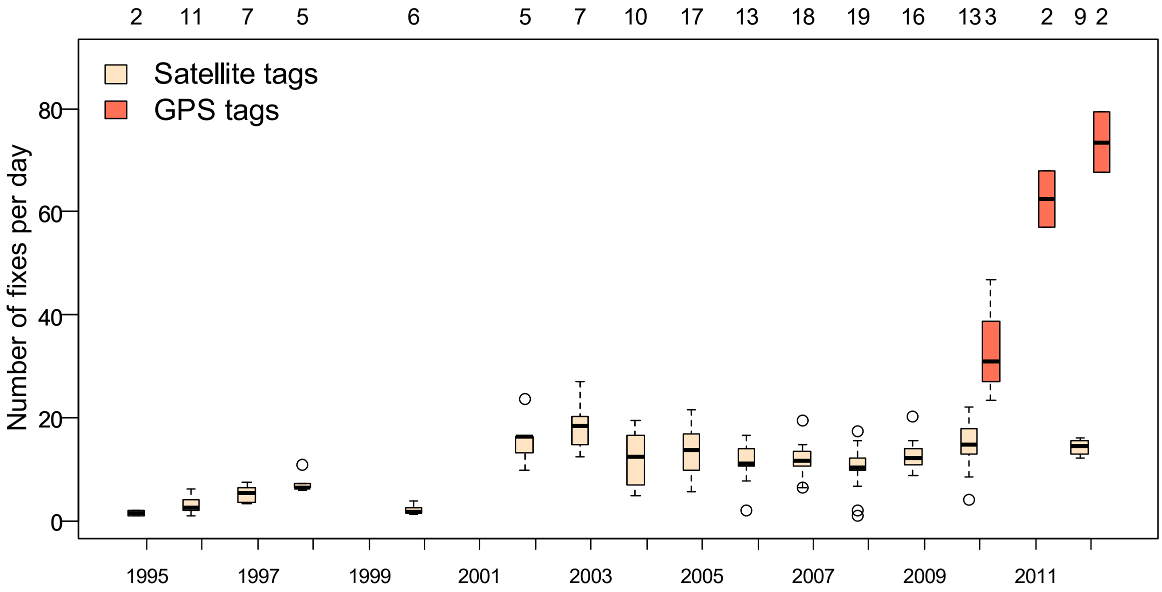

Figure A1-3: Average number of fixes per day by year for *Argos* satellite tags and GPS tags. Numbers on the top axis are the number of tag deployments.

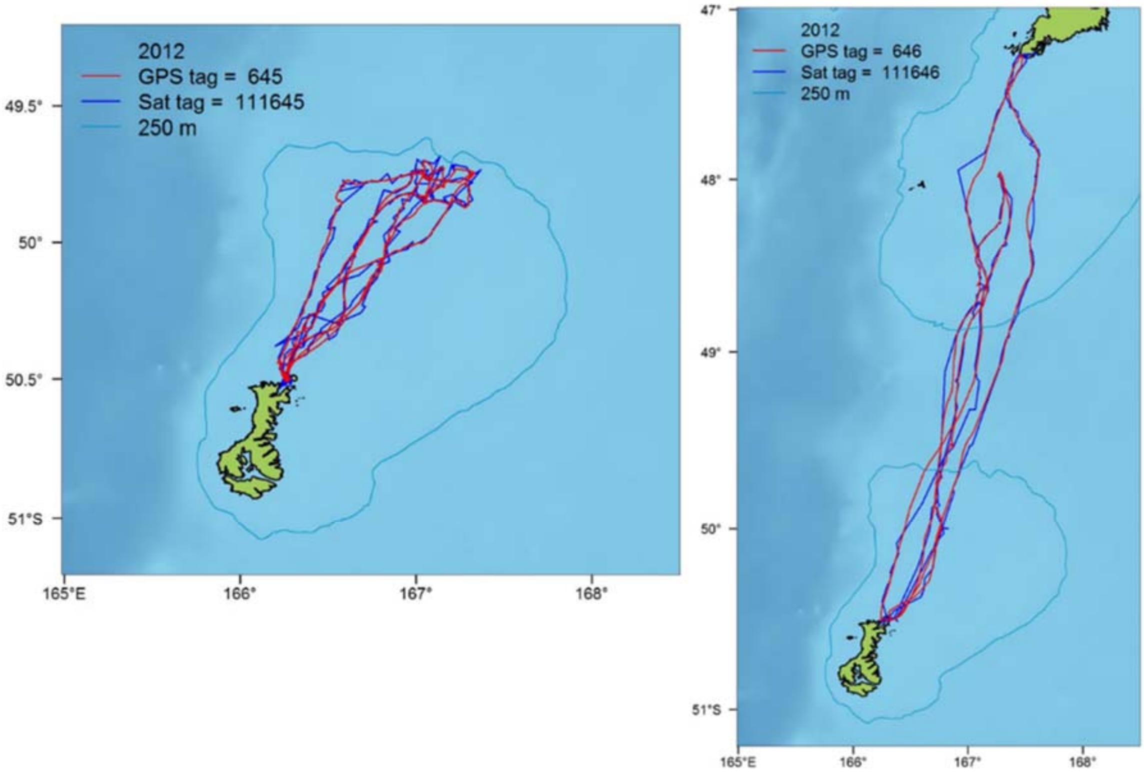

Figure A1-4: Comparison of *Argos* satellite (Sat) tag and GPS tag tracks for two sea lions that were fitted with both tag types in 2012. Groomed and filtered data.

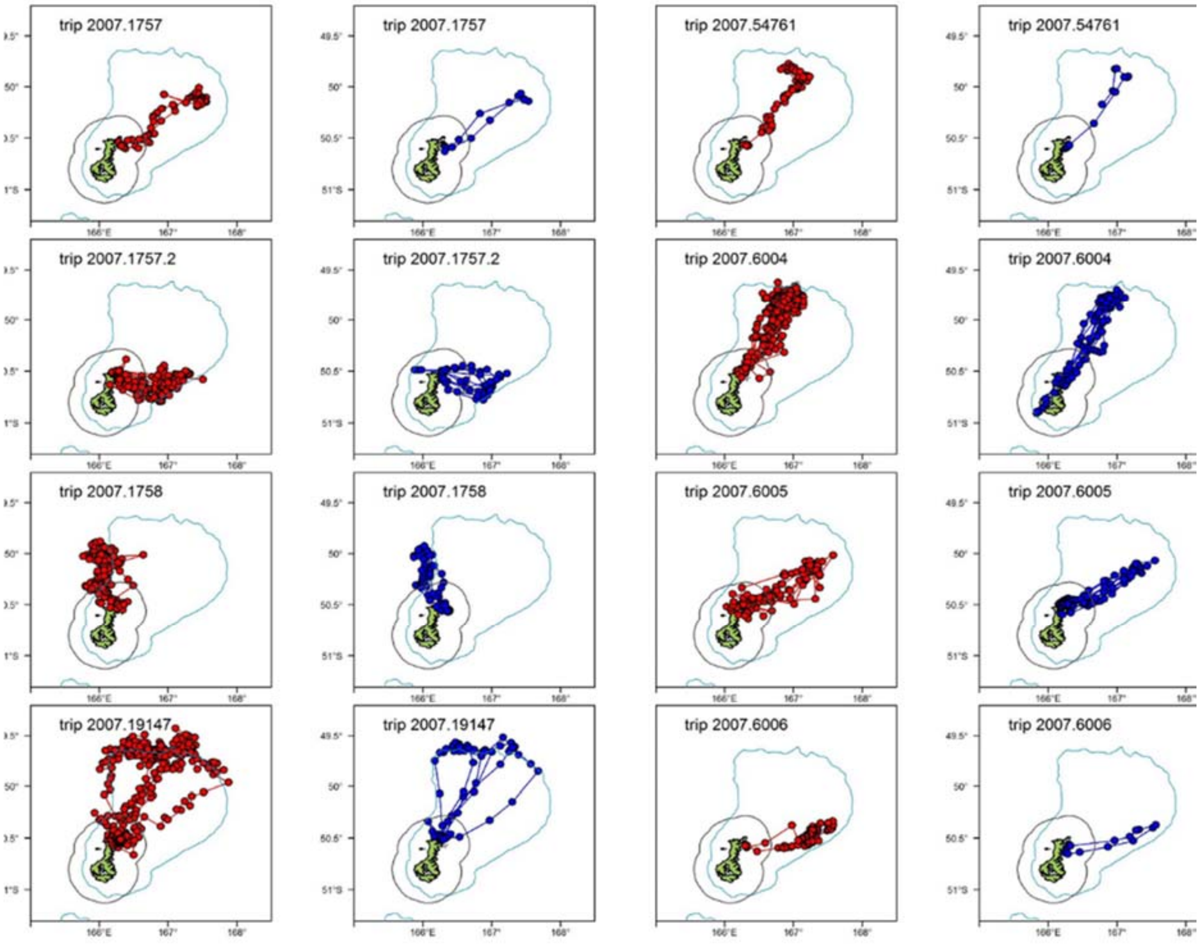

Figure A1-5: Comparison of pairs of groomed, filtered fixes (red symbols) and SSSM fitted locations (blue symbols) for eight selected tagged sea lions. Legends provide the year of tagging (2007 in all cases) and tag number. The 250 m isobath (light blue line) and 12 nautical mile zone (grey line) are also shown.

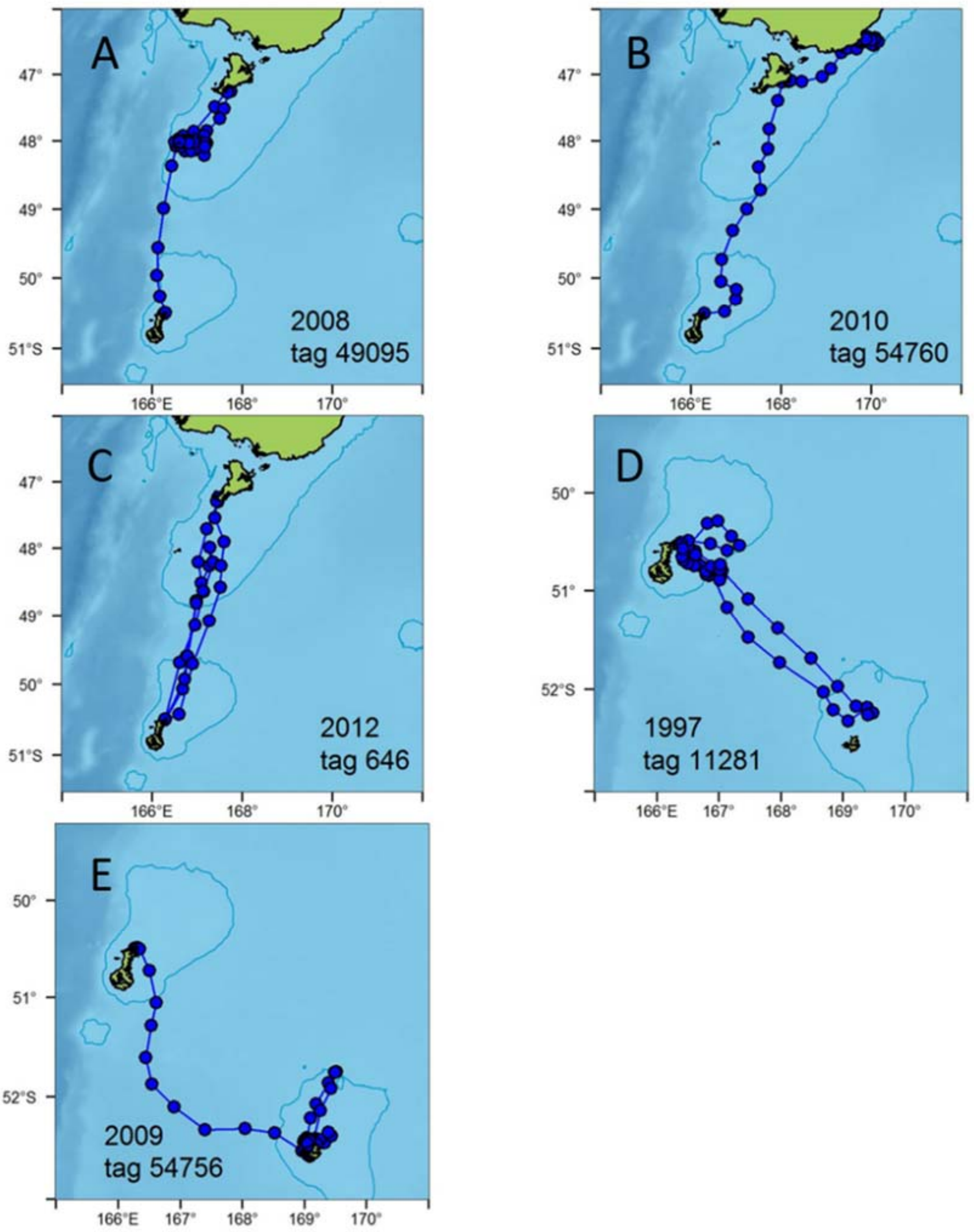

Figure A1-6: Five long-distance sea lion SSSM tracks. A) juvenile male; B) juvenile male; C) lactating female; D) lactating female; E) adult male.

APPENDIX 2 ADDITIONAL PLOTS FROM THE PREDICTION OF NEW ZEALAND SEA LION SPATIAL DENSITY

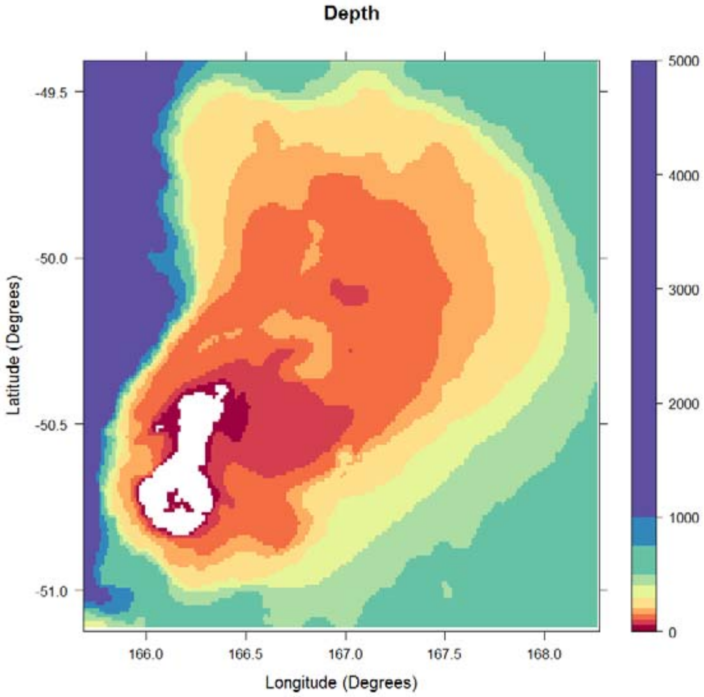

Figure A2-1: Spatial bathymetry layer used for spatial density prediction (NIWA unpublished data).

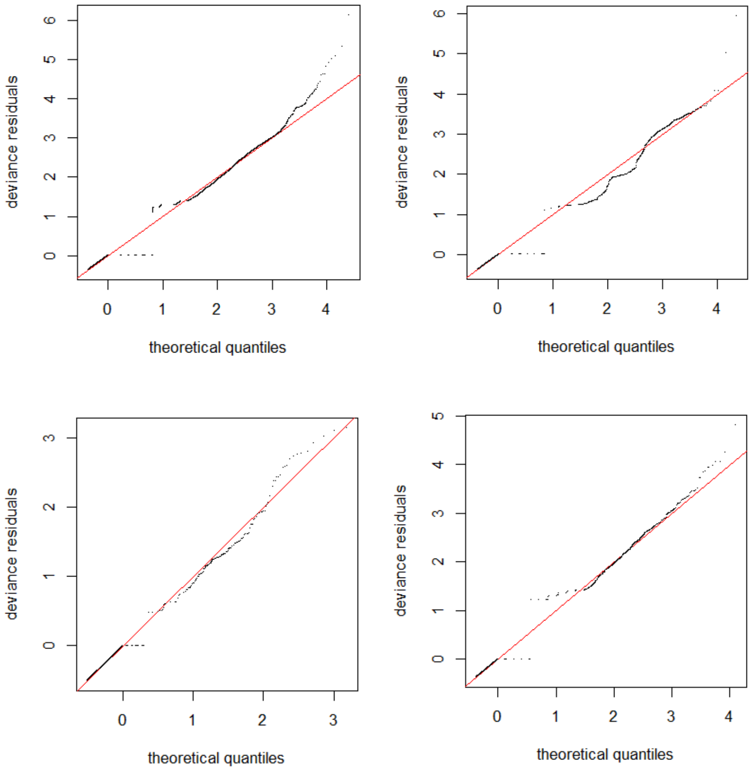

Figure A2-2: Quantile-quantile plots for generalised additive models fitted to spatial density observations of female New Zealand sea lions instrumented at the Auckland Islands: lactating females at Sandy Bay, Enderby Island (top-left); lactating females at Dundas Island (top-left); lactating females at Figure of Eight Island (bottom-left); and juveniles at Sandy Bay, Enderby Island (bottom-right).

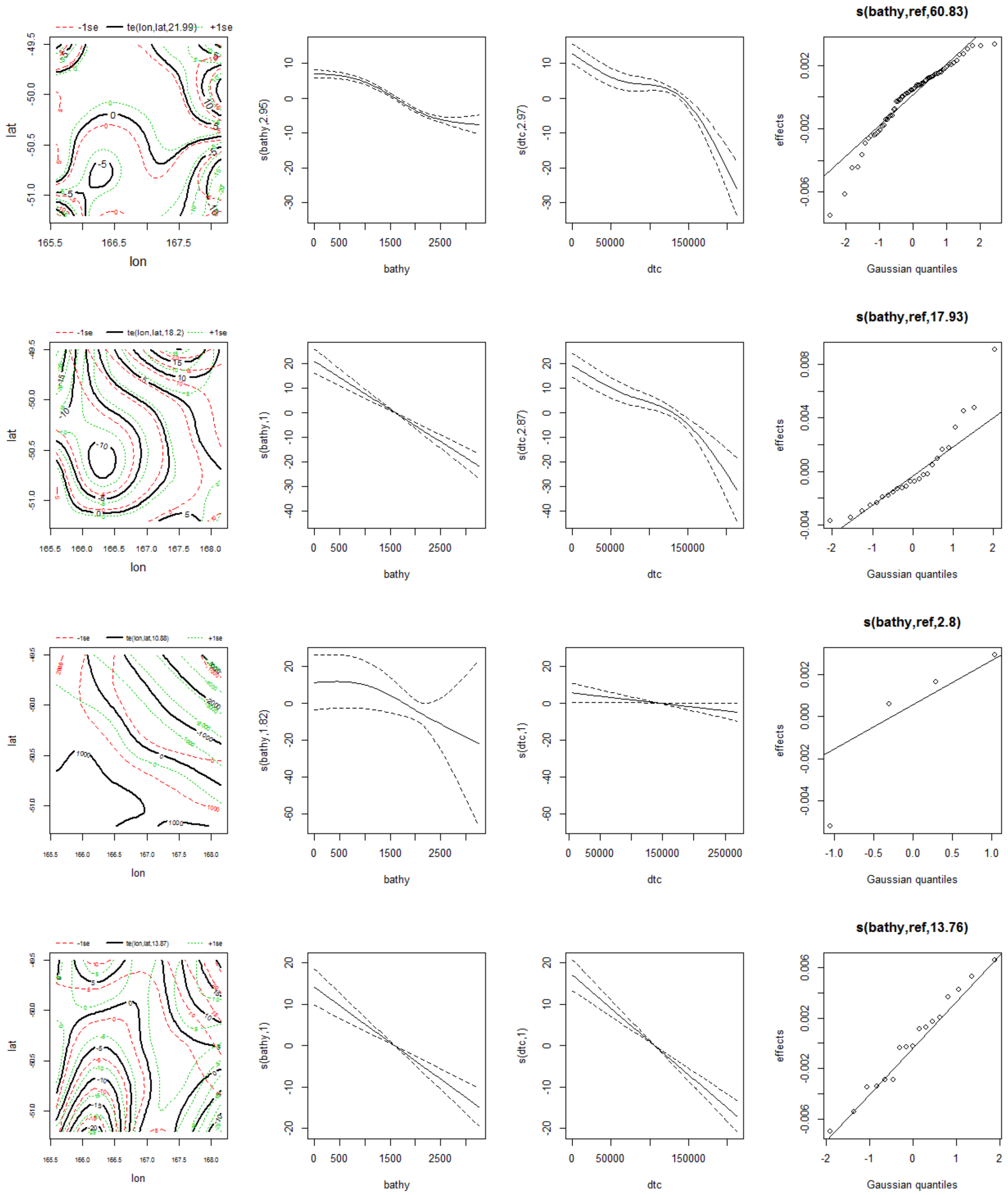

Figure A2-3: Spline plots for generalised additive models fitted to spatial density observations of female New Zealand sea lions instrumented at the Auckland Islands (from top to bottom): lactating females at Sandy Bay, Enderby Island; lactating females at Dundas Island; lactating females at Figure of Eight Island; and juveniles at Sandy Bay, Enderby Island.

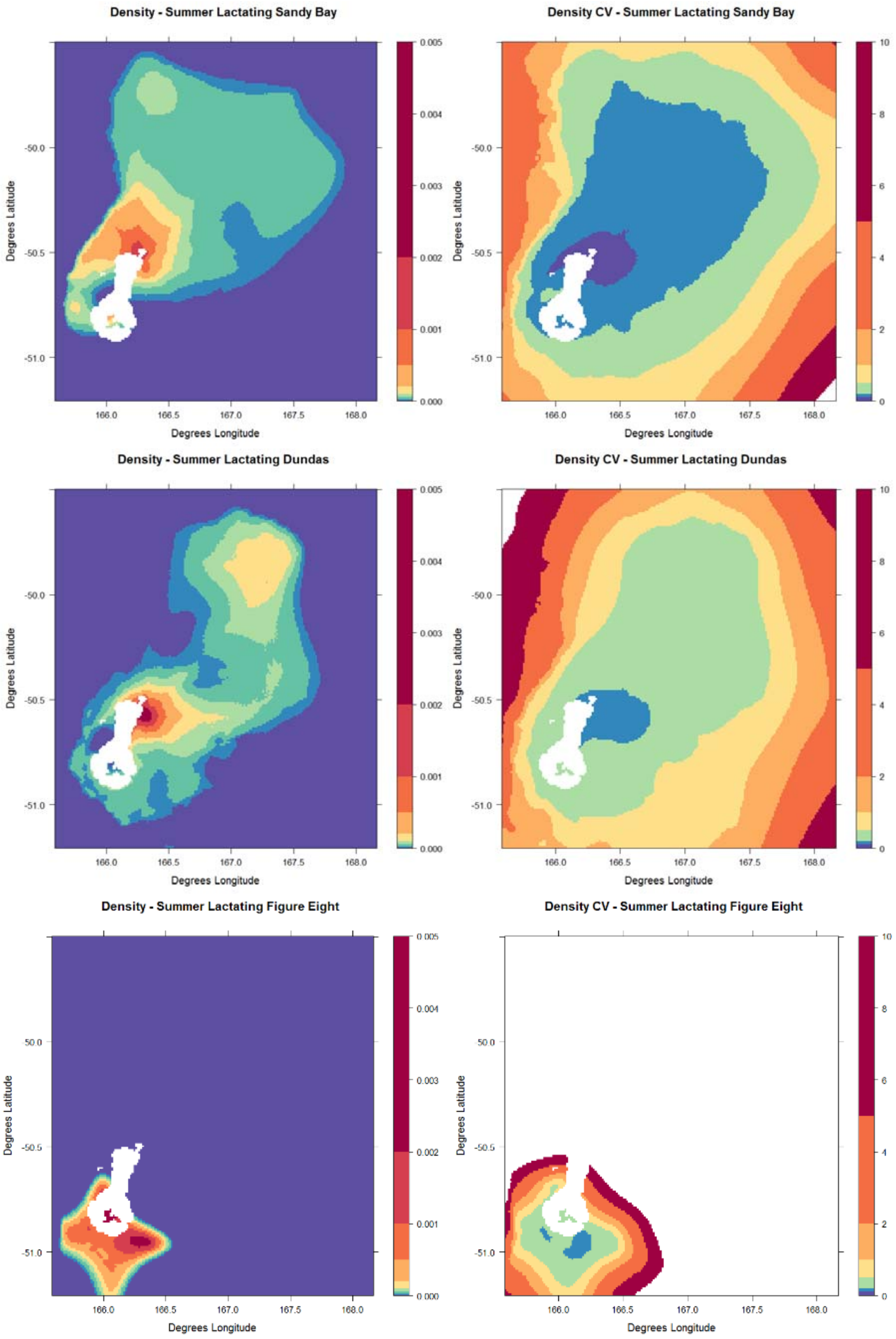

Figure A2-4: Estimated spatial density (left) and coefficients of variation (CVs) of the estimated density (right) for lactating female New Zealand sea lions instrumented at Sandy Bay, Enderby Island (top), Dundas Island (middle row), and Figure of Eight Island (bottom). Note that regions in the Figure of Eight CV plot with CVs greater than 10 were coloured white.

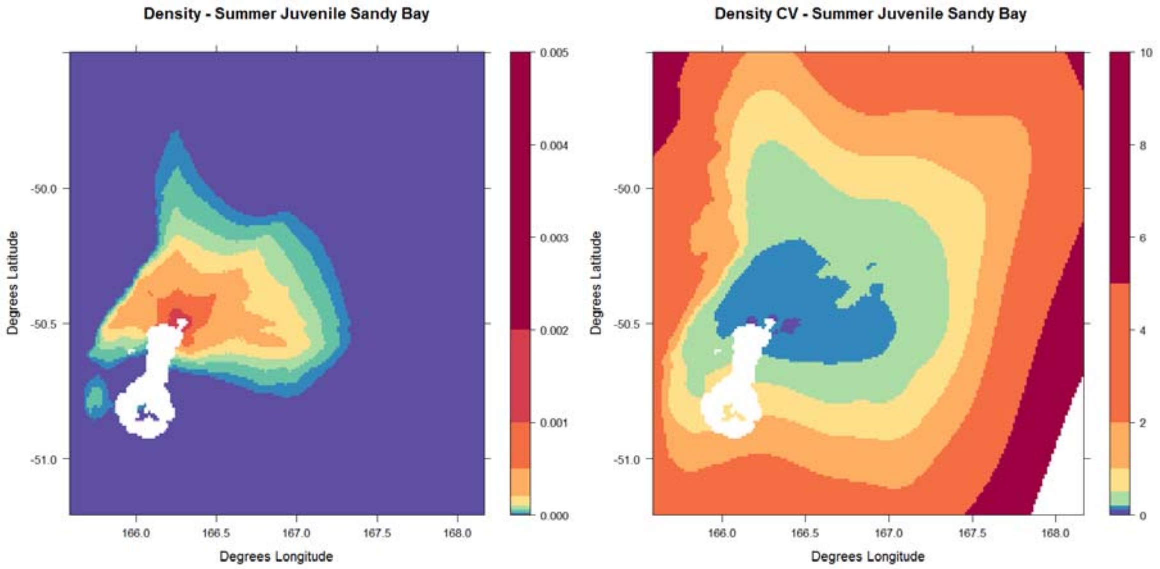

Figure A2-5: Estimated spatial density (left) and coefficients of variation (CVs) of the estimated density (right) for juvenile female New Zealand sea lions instrumented at Sandy Bay, Enderby Island.

APPENDIX 3 SUMMARY OF DATA USED BY SPATIAL RISK MODEL

Table A3-1. Summary of Auckland Islands fishery data used by SEFRA models: bottom trawls targeting southern arrow squid (SQU$_{\text{BT}}$). Note that risk models were fitted to female captures only, and male captures are presented for completeness only.

							Captures on observed trawl events					
Fishing year	All trawl events by fishery group			Observed trawl events by fishery group			Female captures by fishery group			Status of female captures		Male captures
	No SLED	Non-stand. SLED	Stand. SLED	No SLED	Non-stand. SLED	Stand. SLED	No SLED	Non-stand. SLED	Stand. SLED	Alive	Dead	
1992/93	86	0	0	9	–	–	0	–	–	0	0	0
1993/94	1 446	0	0	211	–	–	2	–	–	1	1	1
1994/95	1 375	0	0	110	–	–	1	–	–	0	1	2
1995/96	721	0	0	0	–	–	0	–	–	0	0	0
1996/97	1 544	0	0	192	–	–	2	–	–	0	2	7
1997/98	242	0	0	47	–	–	2	–	–	0	2	2
1998/99	89	0	0	29	–	–	1	–	–	0	1	0
1999/00	455	0	0	66	–	–	1	–	–	0	1	0
2000/01	9	164	0	8	163	–	0	6	–	0	6	4
2001/02	453	45	0	71	35	–	2	0	–	0	2	0
2002/03	111	627	0	107	144	–	0	2	–	0	2	1
2003/04	76	1 376	0	57	196	–	1	2	–	0	3	1
2004/05	46	1 329	0	34	258	–	0	5	–	0	5	2
2005/06	20	1 885	0	8	236	–	2	1	–	0	3	0
2006/07	2	730	0	0	313	–	0	2	–	0	2	1
2007/08	2	632	0	0	274	–	0	2	–	0	2	2
2008/09	0	0	1 068	–	–	365	–	–	1	0	1	1
2009/10	0	0	1 026	–	–	236	–	–	2	1	1	0
2010/11	0	0	1 218	–	–	367	–	–	0	0	0	0
2011/12	0	0	973	–	–	331	–	–	0	0	0	0
2012/13	0	0	813	–	–	671	–	–	3	0	3	0
2013/14	0	0	477	–	–	396	–	–	2	0	2	0
2014/15	0	0	328	–	–	302	–	–	0	0	0	0
2015/16	0	0	822	–	–	718	–	–	0	0	0	0
2016/17	0	0	1 074	–	–	733	–	–	2	0	2	0

Table A3-2. Summary of Auckland Islands fishery data used by SEFRA models: midwater trawls targeting southern arrow squid (SQU_{MW}). Note that risk models were fitted to female captures only, and male captures are presented for completeness only.

Fishing year	All trawl events by fishery group			Observed trawl events by fishery group			Captures on observed trawl events: Female captures by fishery group			Status of female captures		Male captures
	No SLED	Non-stand. SLED	Stand. SLED	No SLED	Non-stand. SLED	Stand. SLED	No SLED	Non-stand. SLED	Stand. SLED	Alive	Dead	
1992/93	568	0	0	188	–	–	3	–	–	0	3	2
1993/94	3 226	0	0	211	–	–	0	–	–	0	0	1
1994/95	2 633	0	0	172	–	–	3	–	–	0	3	2
1995/96	3 747	0	0	557	–	–	10	–	–	0	10	3
1996/97	2 177	0	0	543	–	–	7	–	–	0	7	12
1997/98	1 219	0	0	289	–	–	2	–	–	0	2	9
1998/99	313	0	0	127	–	–	3	–	–	0	3	1
1999/00	751	0	0	372	–	–	12	–	–	0	12	12
2000/01	24	386	0	24	383	–	1	15	–	0	16	13
2001/02	168	981	0	120	337	–	4	8	–	0	12	7
2002/03	92	636	0	45	120	–	2	3	–	0	5	3
2003/04	33	1 109	0	15	525	–	1	10	–	0	11	1
2004/05	17	1 301	0	6	508	–	0	0	–	0	0	2
2005/06	32	522	0	16	290	–	2	5	–	0	7	0
2006/07	23	562	0	13	210	–	0	4	–	0	4	0
2007/08	12	619	0	8	309	–	0	1	–	0	1	0
2008/09	0	0	857	–	–	398	–	–	0	0	0	0
2009/10	0	0	162	–	–	67	–	–	1	0	1	0
2010/11	0	0	365	–	–	180	–	–	0	0	0	0
2011/12	0	0	308	–	–	240	–	–	0	0	0	0
2012/13	0	0	214	–	–	214	–	–	0	0	0	0
2013/14	0	0	260	–	–	226	–	–	0	0	0	0
2014/15	0	0	305	–	–	257	–	–	1	0	1	0
2015/16	0	0	543	–	–	543	–	–	0	0	0	0
2016/17	0	0	206	–	–	168	–	–	1	0	1	0

Table A3-3. Summary of Auckland Islands fishery data used by SEFRA models: trawls targeting scampi (SCI) and all other species (OTH). Note that risk models were fitted to female captures only, and male captures are presented for completeness only.

Fishing year	All trawl events by fishery group		Observed trawl events by fishery group		Live female captures on observed trawl events		Dead female captures on observed trawl events		Male captures on observed trawl events	
	SCI	OTH	SCI	OTH	SCI	OTH	SCI	OTH	SCI	OTH
1992/93	835	195	149	36	0	0	2	0	1	0
1993/94	1 314	308	272	23	0	0	0	0	0	0
1994/95	1 349	492	50	33	0	0	0	0	0	0
1995/96	1 312	411	67	25	2	0	0	1	0	0
1996/97	1 227	296	200	13	1	0	0	0	0	0
1997/98	1 109	688	137	118	0	0	0	0	0	0
1998/99	1 255	525	23	55	0	0	0	0	0	0
1999/00	1 383	751	74	98	0	0	0	0	0	0
2000/01	1 419	577	84	42	2	0	2	0	0	0
2001/02	1 603	590	154	23	0	0	0	0	0	0
2002/03	1 351	543	150	70	0	0	0	1	0	0
2003/04	1 363	289	169	49	0	0	3	0	0	0
2004/05	1 275	170	0	12	–	0	–	0	–	0
2005/06	1 331	39	118	6	0	0	1	0	0	0
2006/07	1 328	38	101	2	0	0	0	0	1	0
2007/08	1 327	147	93	66	0	0	0	0	0	0
2008/09	1 457	121	61	60	0	0	1	0	0	0
2009/10	940	77	92	52	0	0	0	0	0	0
2010/11	1 401	131	207	49	0	0	0	0	0	0
2011/12	1 247	57	119	17	0	0	0	0	0	0
2012/13	1 093	60	136	26	0	0	0	0	0	0
2013/14	850	203	52	47	0	0	0	0	0	0
2014/15	548	224	0	69	–	0	–	0	–	0
2015/16	1 414	140	66	36	0	0	0	0	0	0
2016/17	1 677	170	354	86	0	0	0	0	0	0

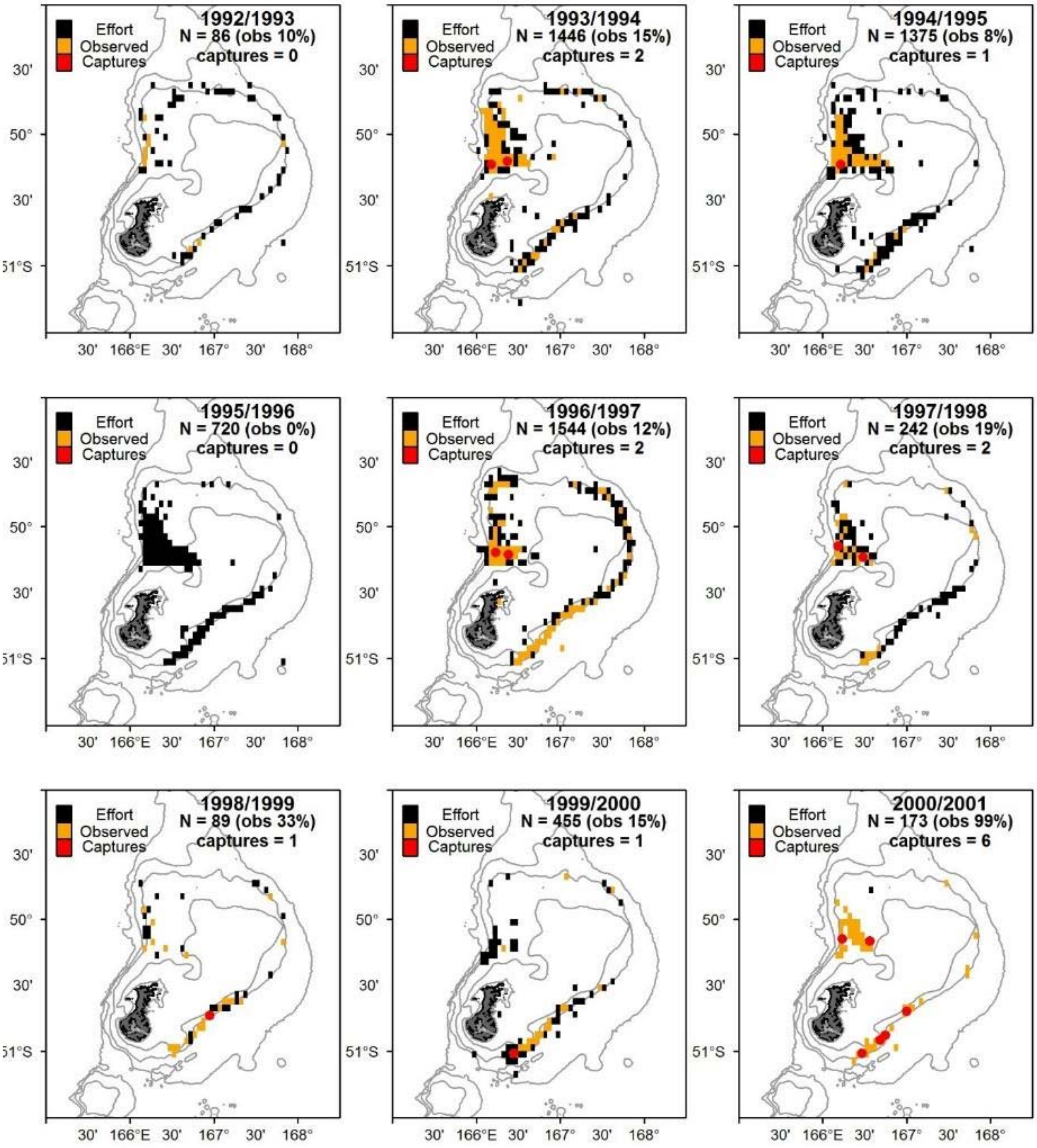

Figure A3-1: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in bottom trawls targeting southern arrow squid (SQU$_{BT}$), 1992/93 to 2000/01.

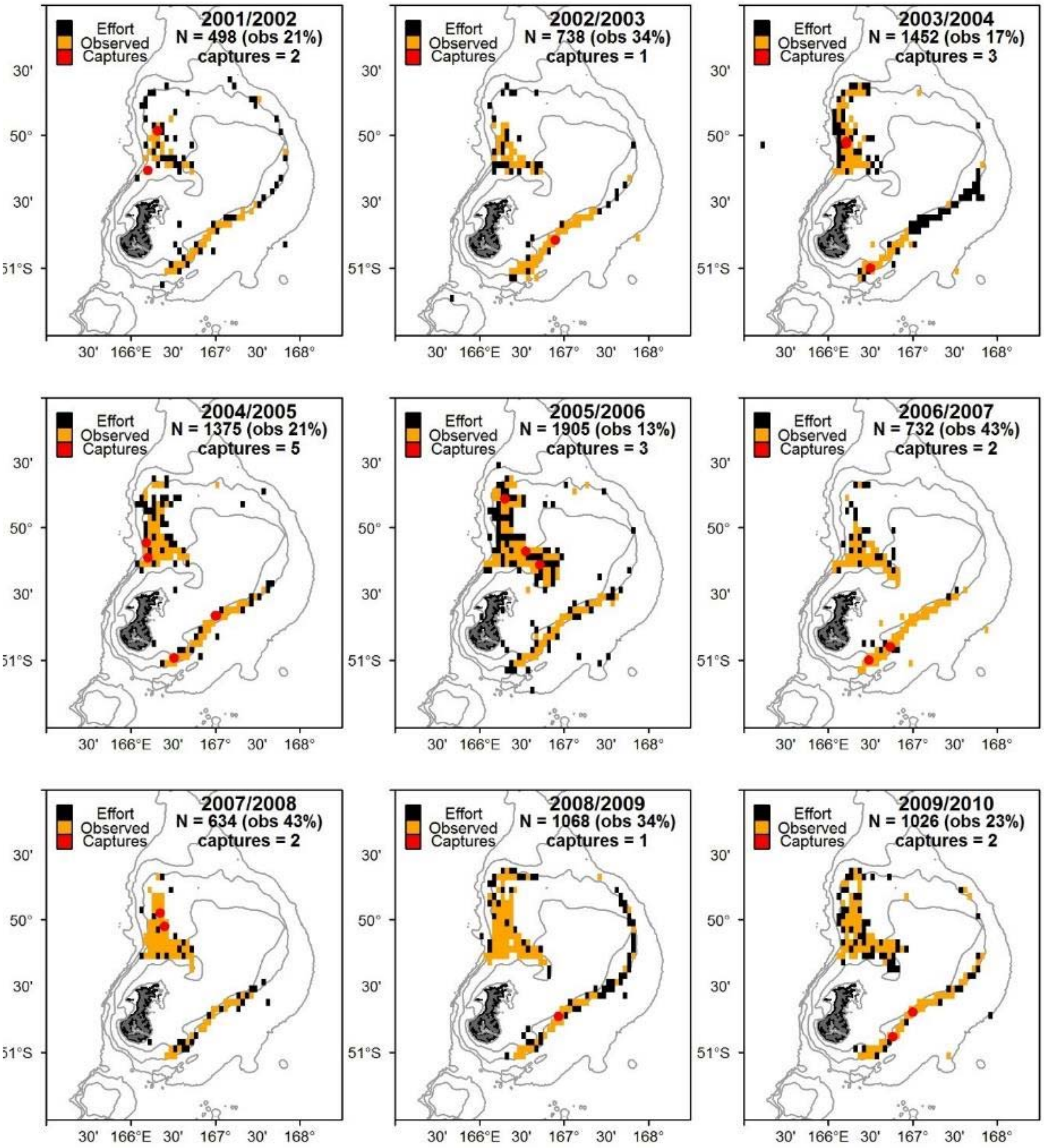

Figure A3-2: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in bottom trawls targeting southern arrow squid (SQU$_{BT}$), 2001/02 to 2009/10.

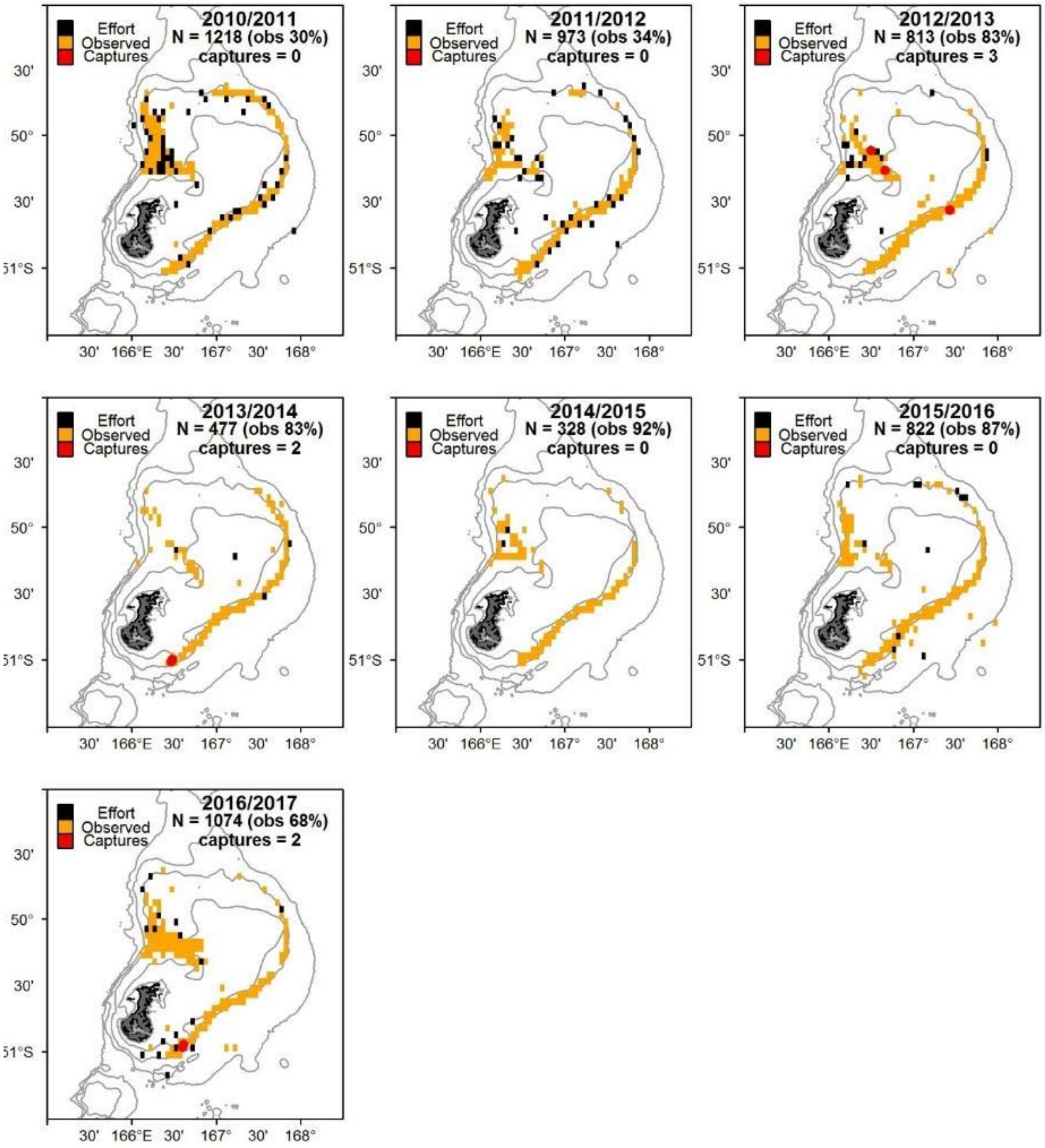

Figure A3-3: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in bottom trawls targeting southern arrow squid (SQU$_{BT}$), 2010/11 to 2016/17.

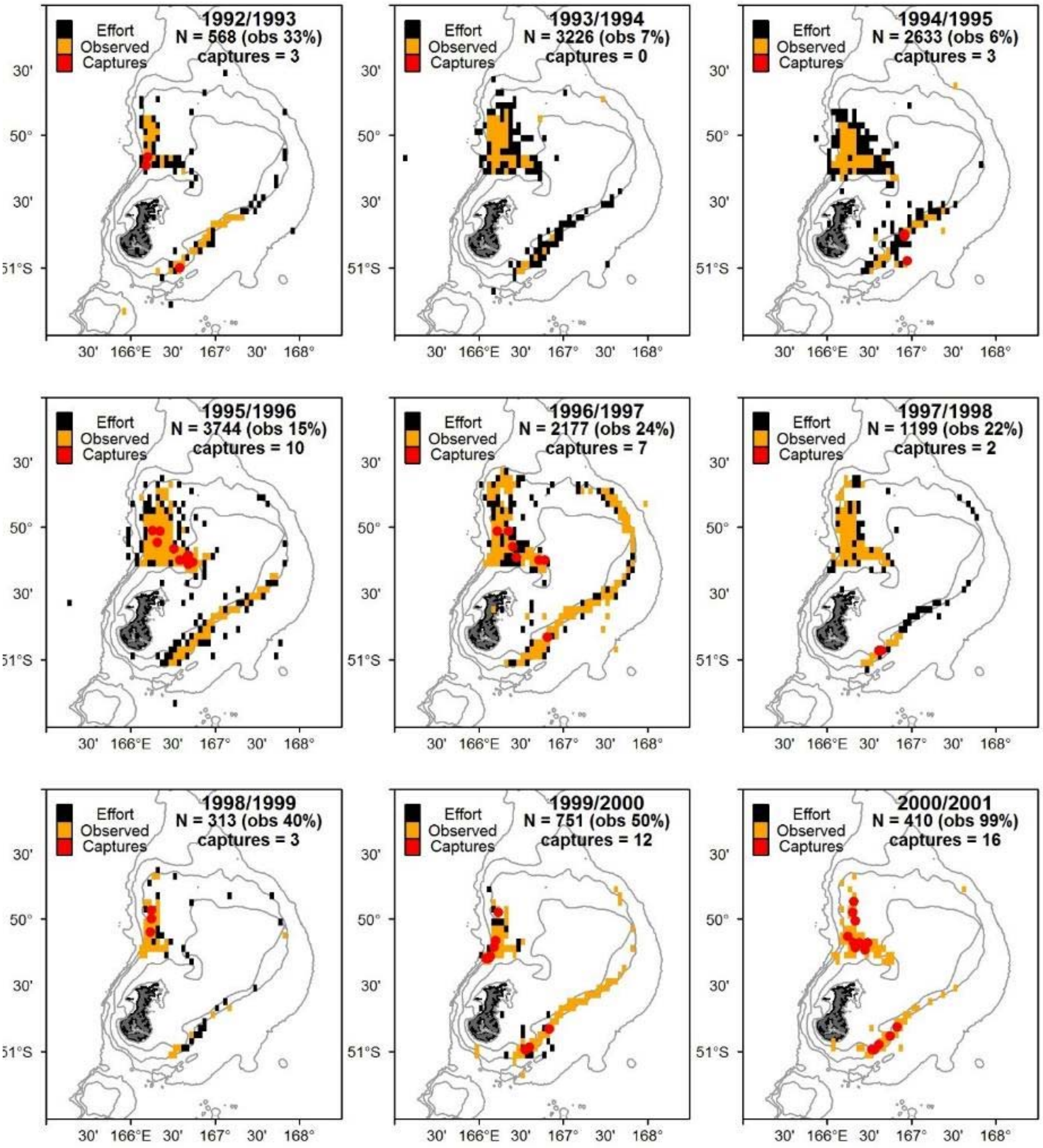

Figure A3-4: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in midwater trawls targeting southern arrow squid ($\mathrm{SQU_{MW}}$), 1992/93 to 2000/01.

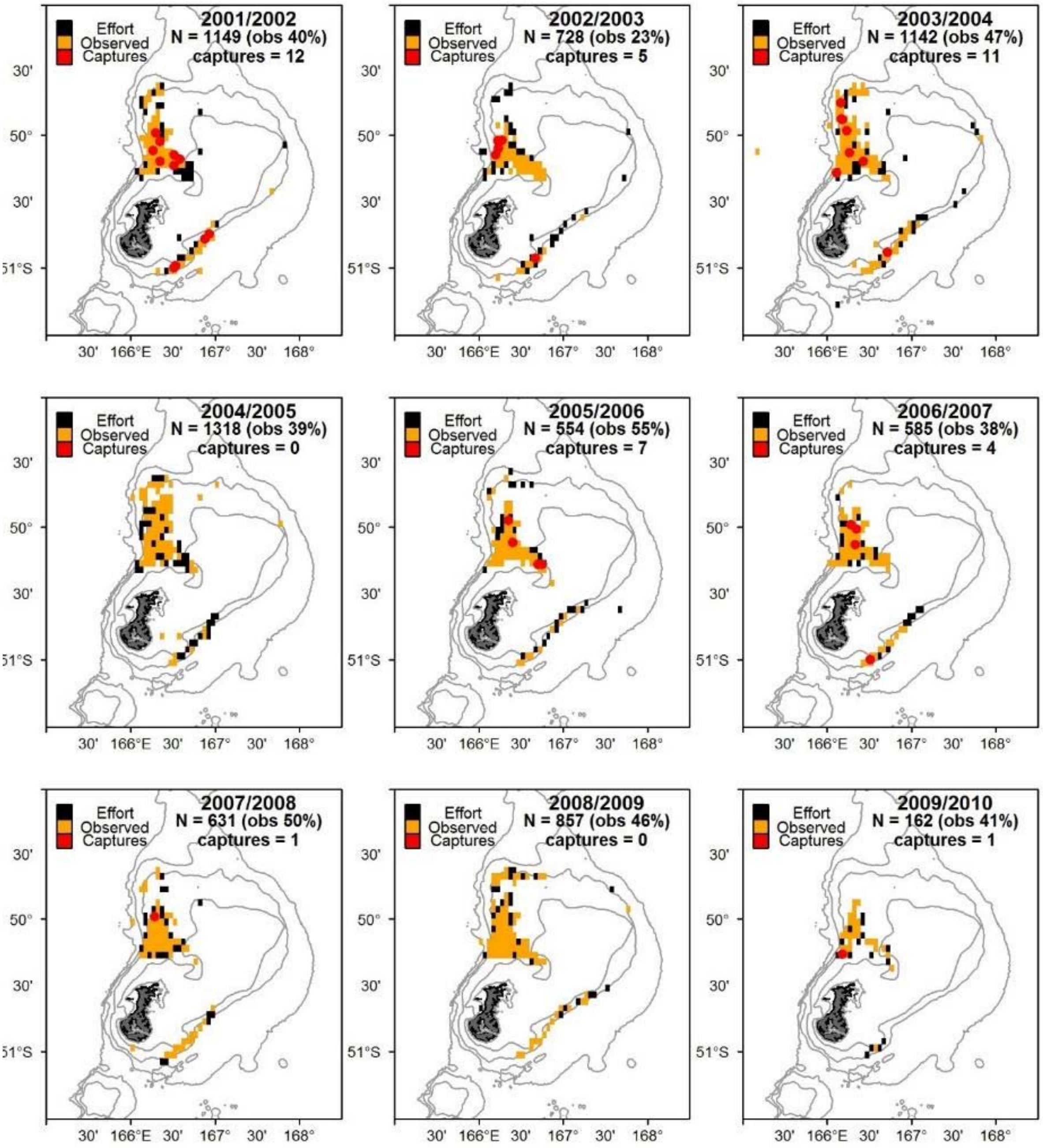

Figure A3-5: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in midwater trawls targeting southern arrow squid (SQU$_{\text{MW}}$), 2001/02 to 2009/10.

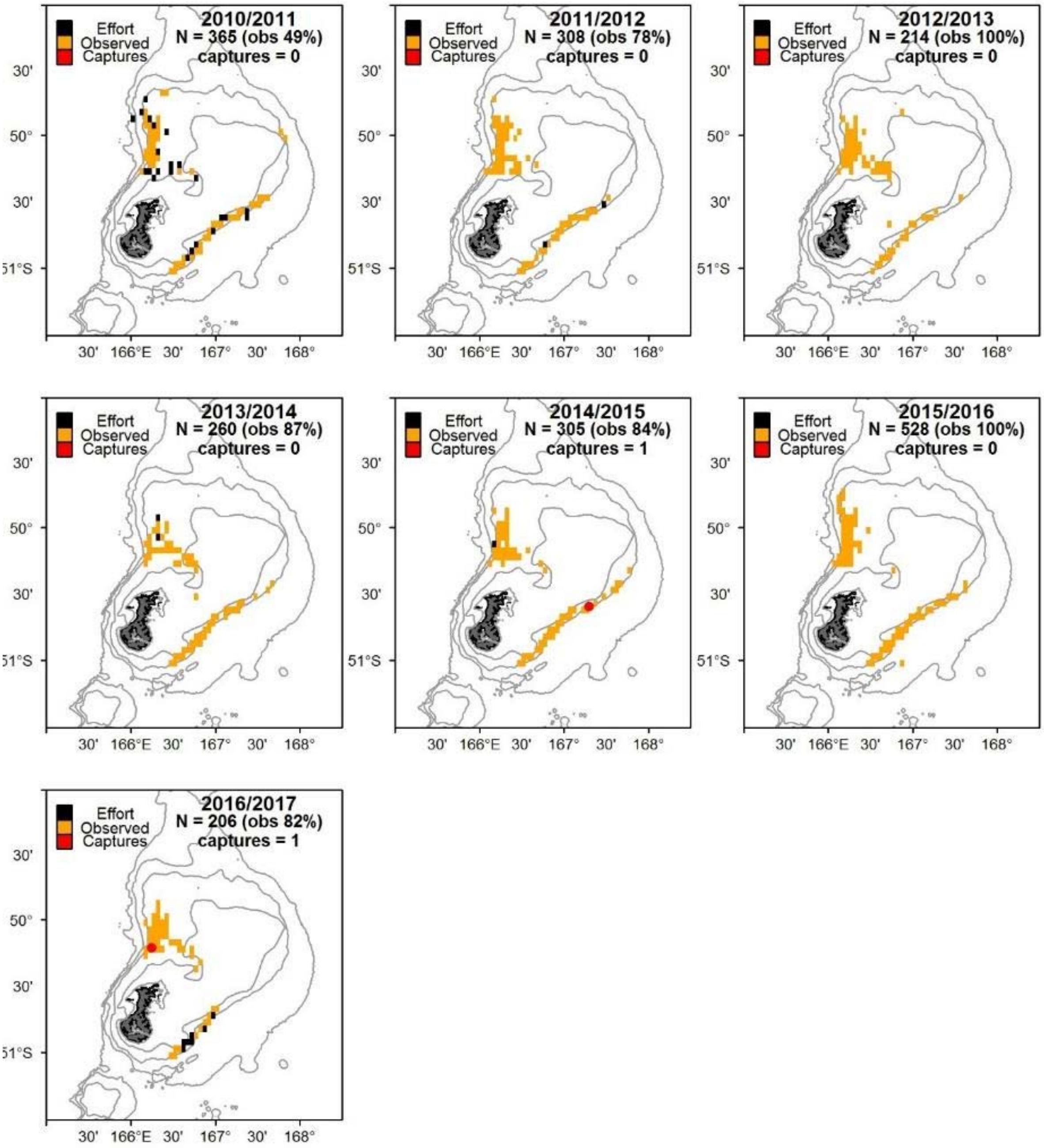

Figure A3-6: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in midwater trawls targeting southern arrow squid (SQU_{MW}), 2010/11 to 2016/17.

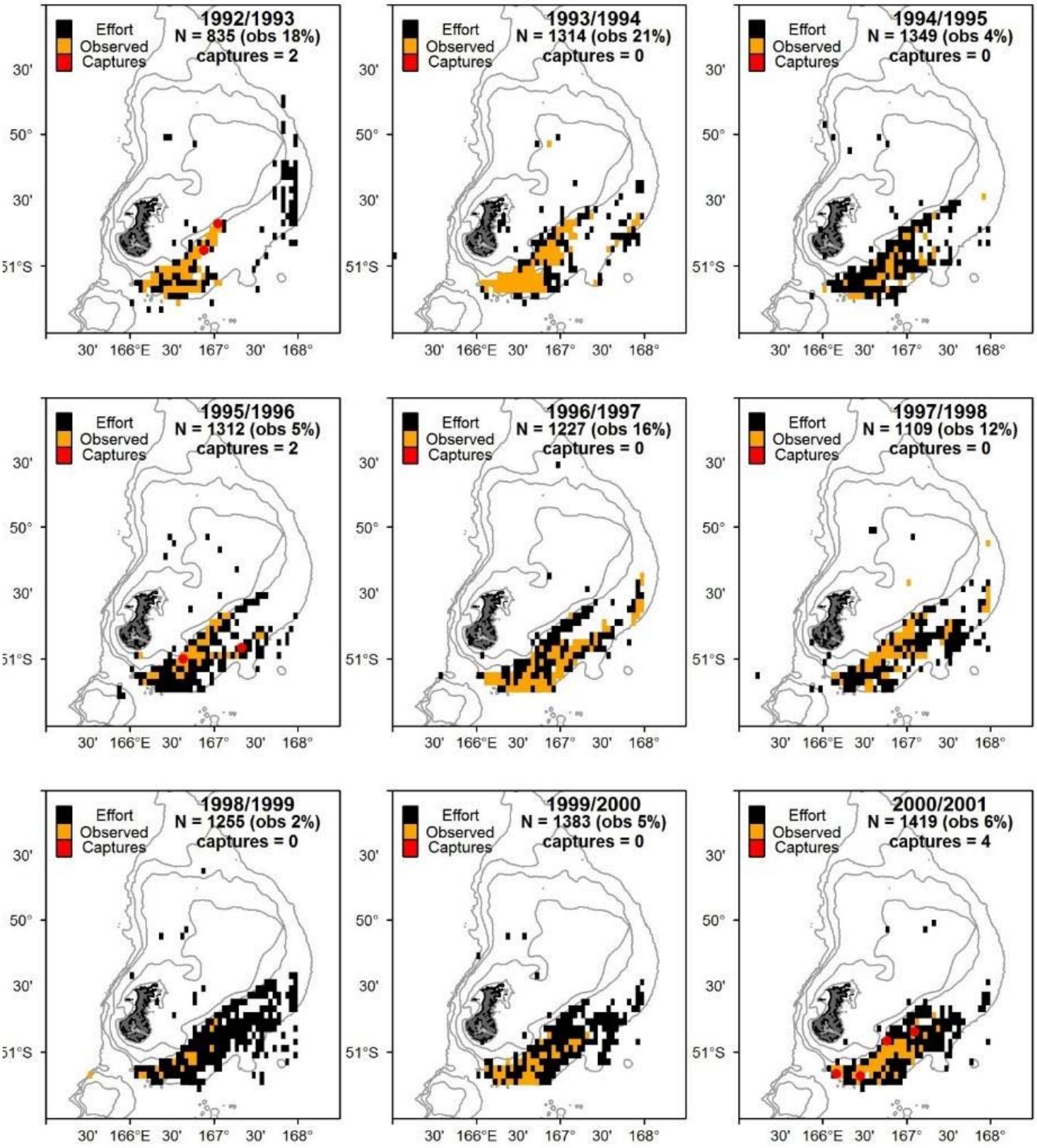

Figure A3-7: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in trawls targeting scampi (SCI), 1992/93 to 2000/01.

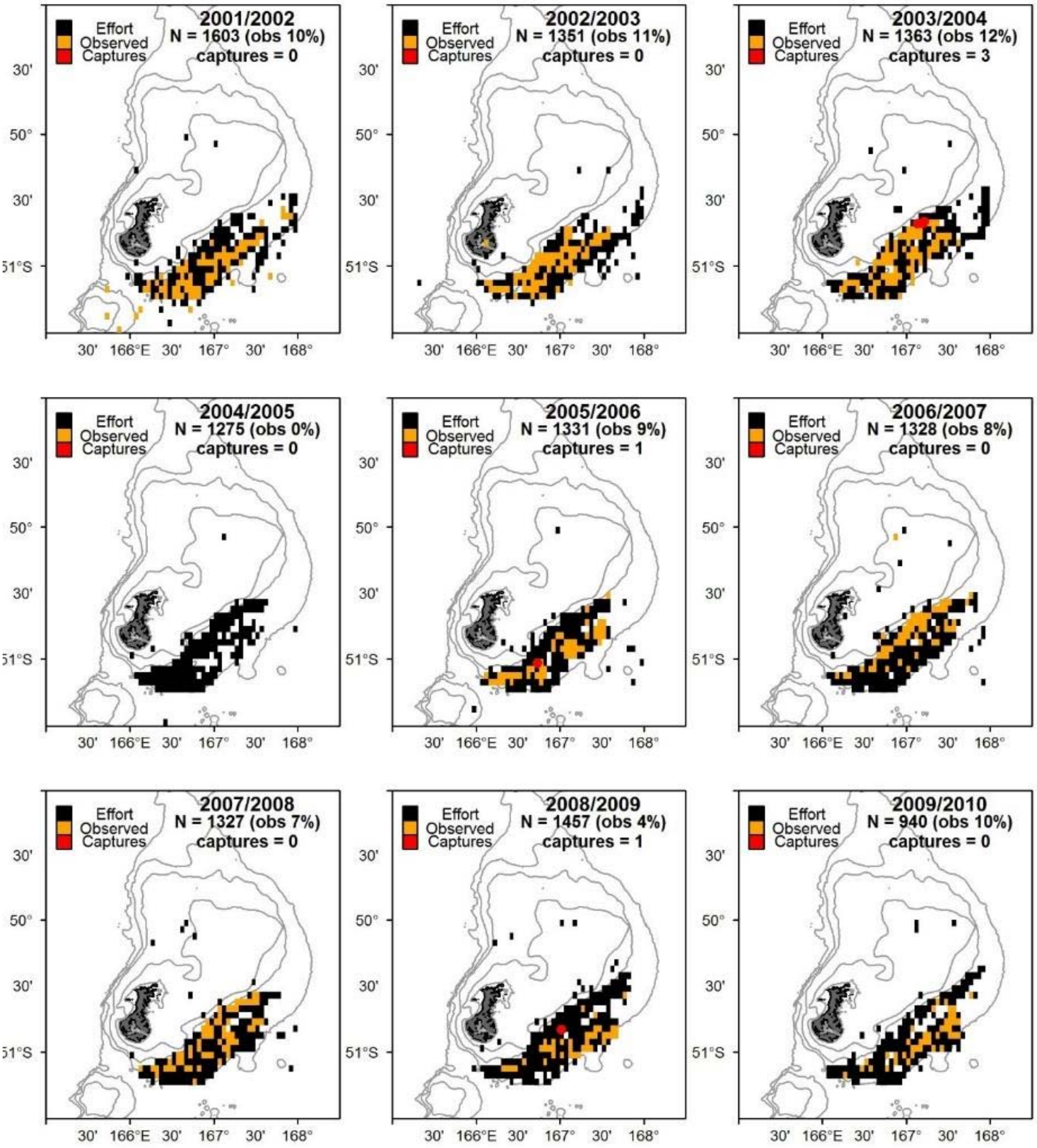

Figure A3-8: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in trawls targeting scampi (SCI), 2001/02 to 2009/10.

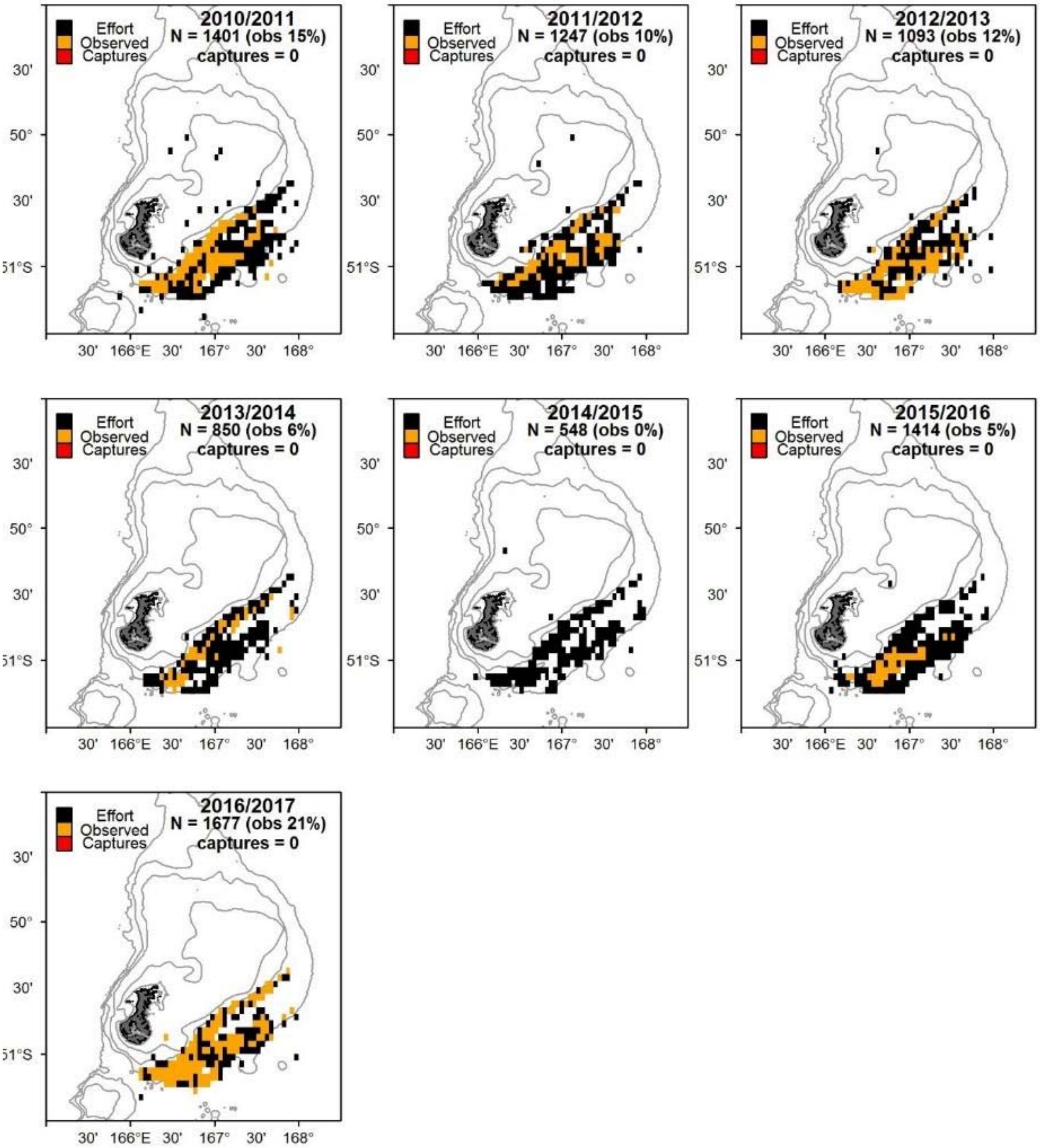

Figure A3-9: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in trawls targeting scampi (SCI), 2010/11 to 2016/17.

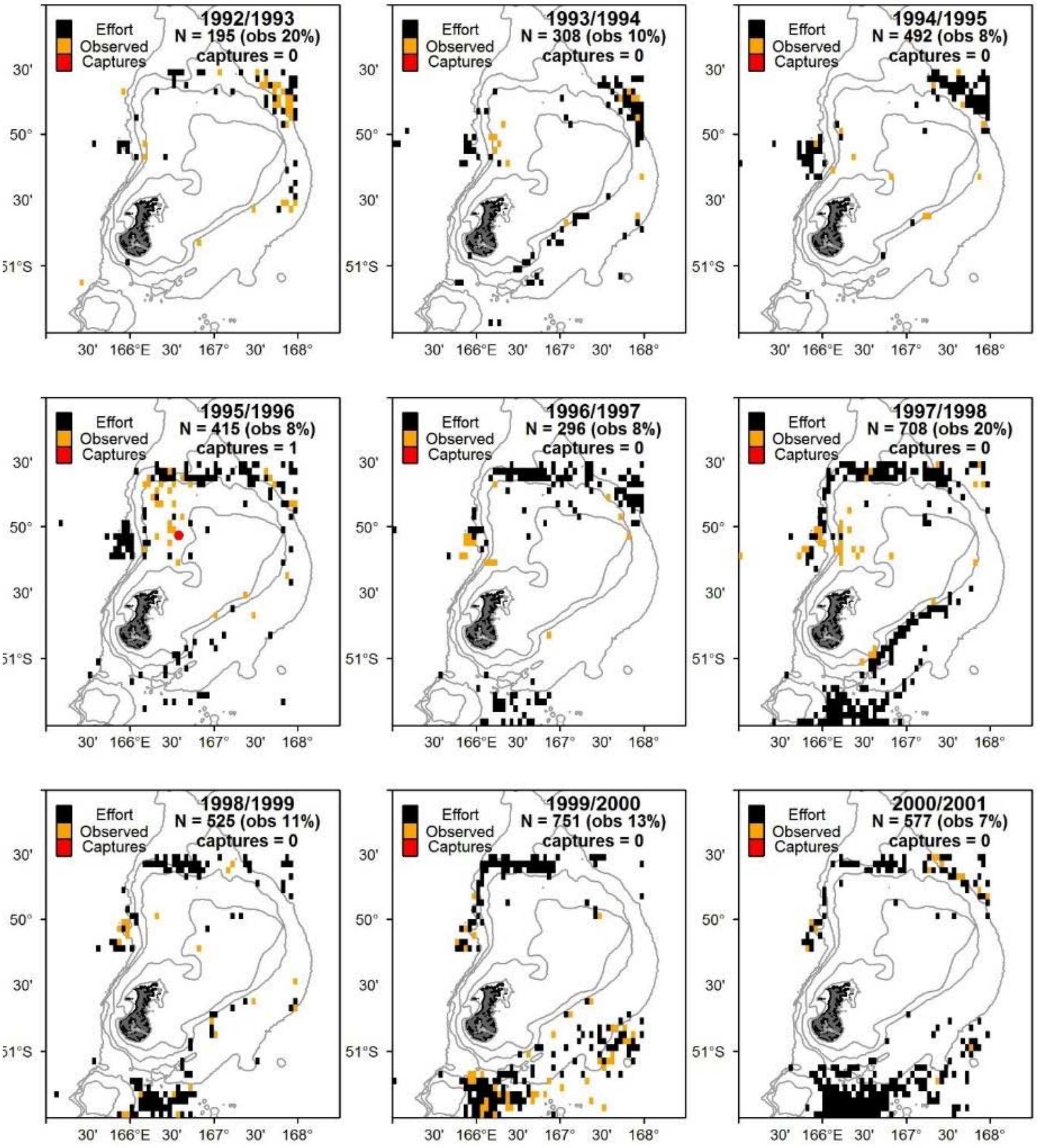

Figure A3-10: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in trawls targeting other species (OTH), 1992/93 to 2000/01.

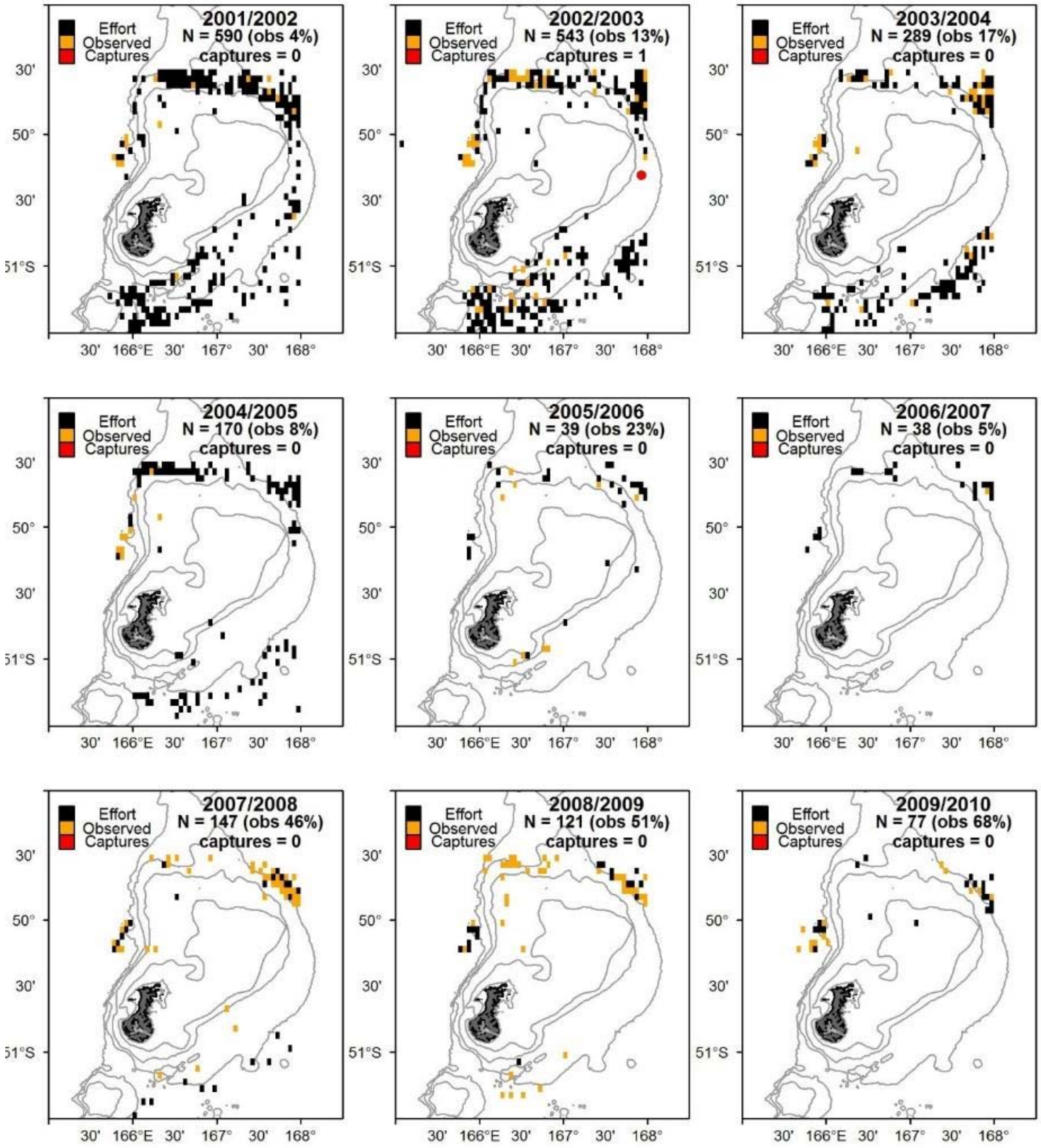

Figure A3-11: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in trawls targeting other species (OTH), 2001/02 to 2009/10.

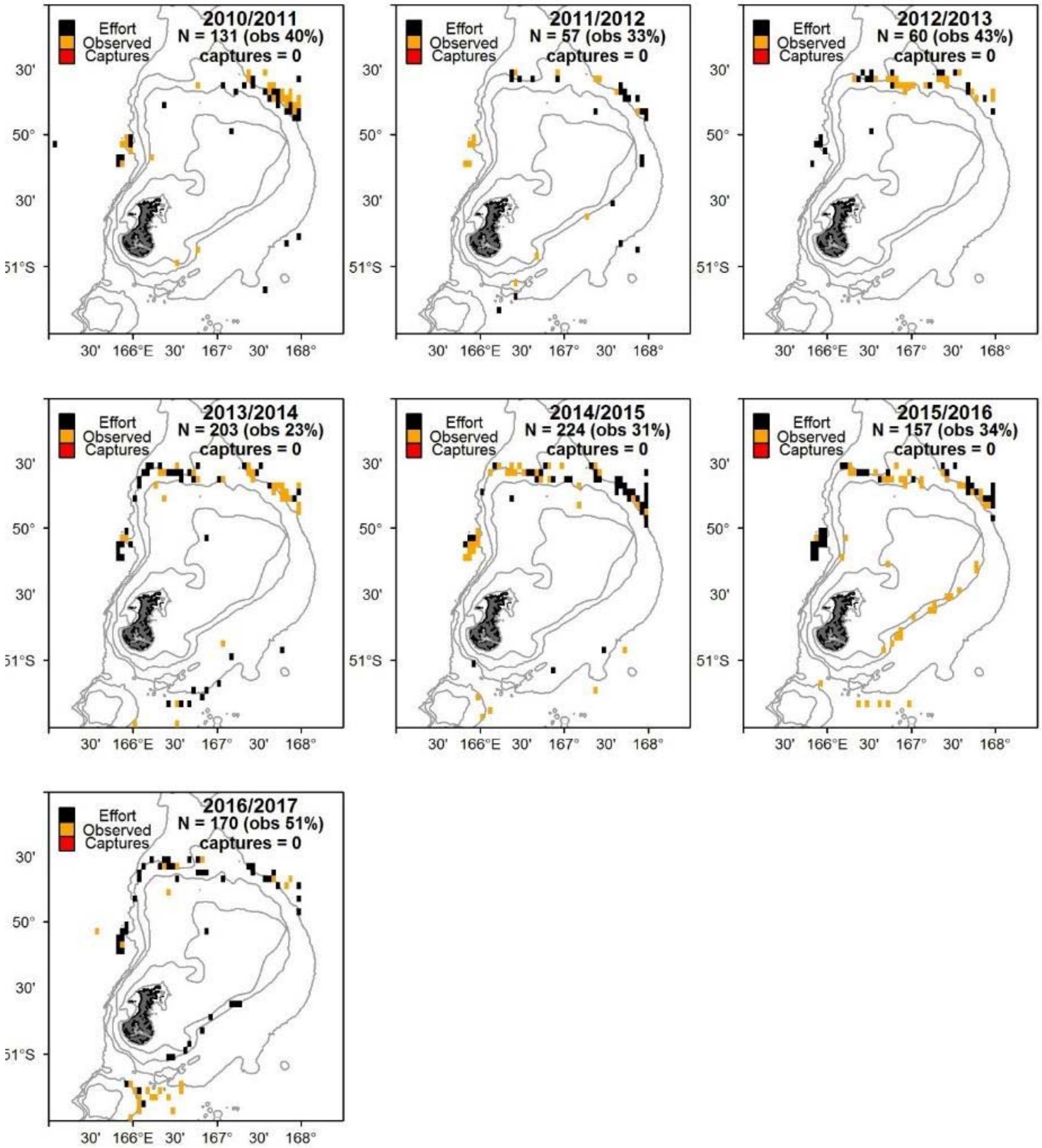

Figure A3-12: Spatial distribution of grid cells with fishing effort, observed effort and observed captures of female New Zealand sea lions in trawls targeting other species (OTH), 2010/11 to 2016/17.

APPENDIX 4 SPATIAL RISK MODEL DIAGNOSTICS

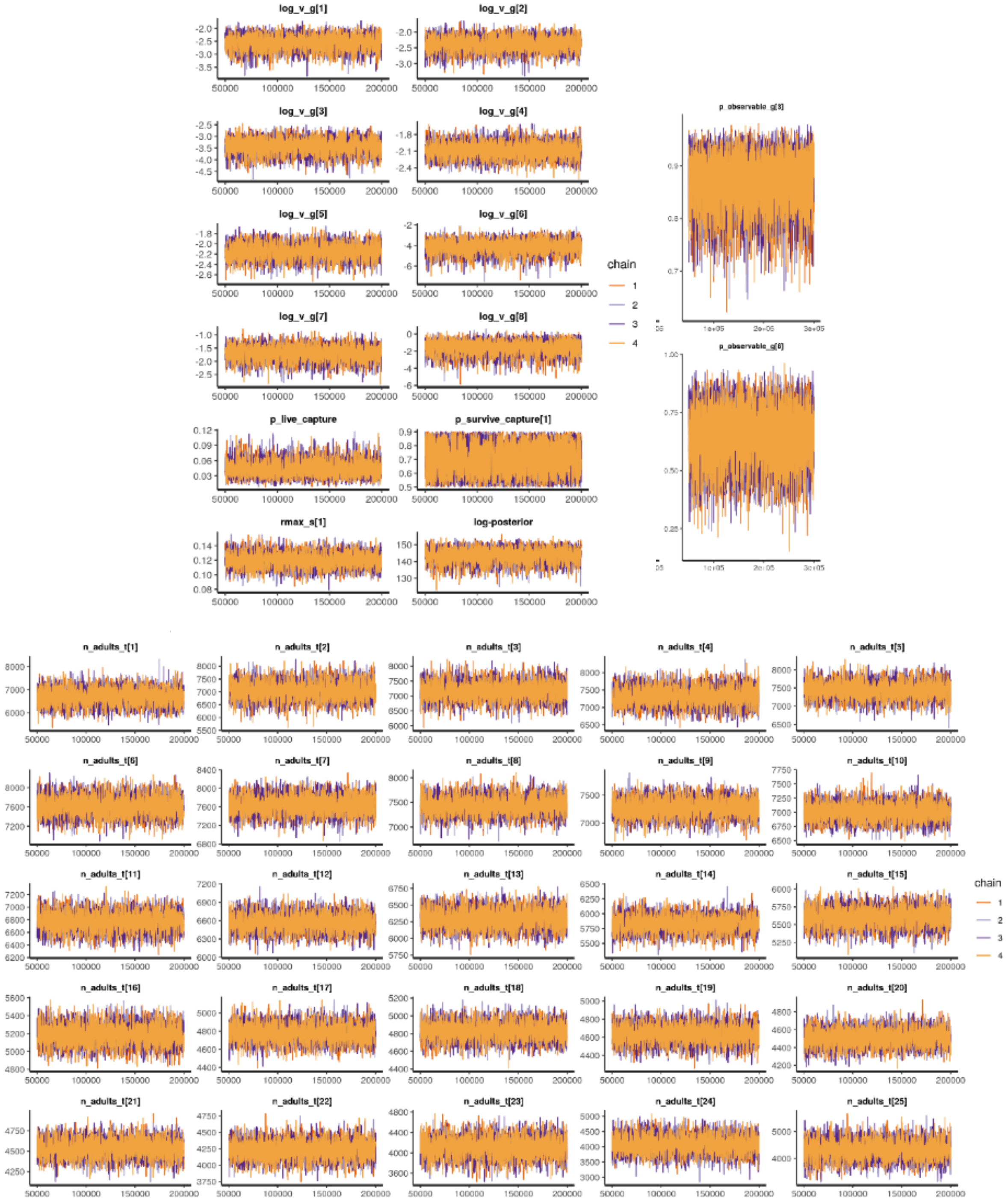

Figure A4-1: MCMC traces for all estimated model parameters, simulated parameters, and log-posterior, for the base case risk model, using $\phi = 0.10$. Different colours were used for each of the four MCMC chains.

Figure A4-2: MCMC priors (pink) and posteriors (blue) for all estimated model parameters and simulated parameters from the base case risk model, using $\phi = 0.10$.

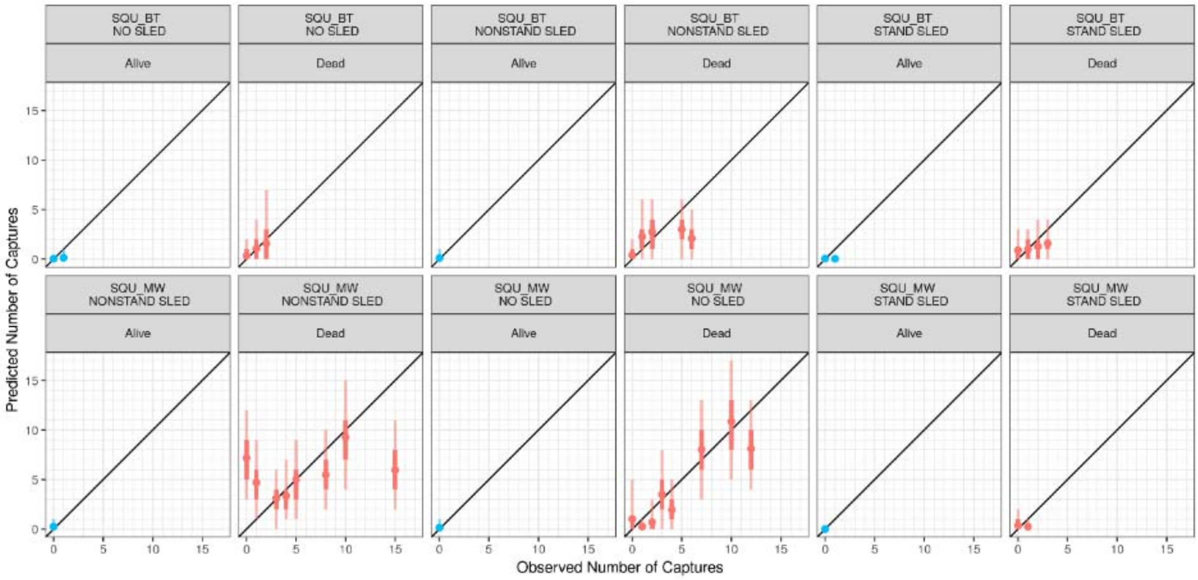

Figure A4-3: MCMC model run fits to annual observed captures for trawl groups targeting southern arrow squid (SQU 6T): (top row) "SQU_BT_NO_SLED" = bottom trawls targeting southern arrow squid without a sea lion exclusion device (SLED), "SQU_BT_NONSTAND_SLED" = bottom trawls targeting squid with a SLED using a non-standardised configuration, "SQU_BT_STAND_SLED" = bottom trawls targeting squid with a SLED using a standardised configuration; (bottom row) midwater trawls, with same labelling used for alternative SLED use groups. A separate plot is shown for each fishery group and for live (blue bars) or dead captures (red bars). Each plot shows annual observed captures estimated by the risk model (point = median, thick bar = interquartile range, whisker = 95% CI)

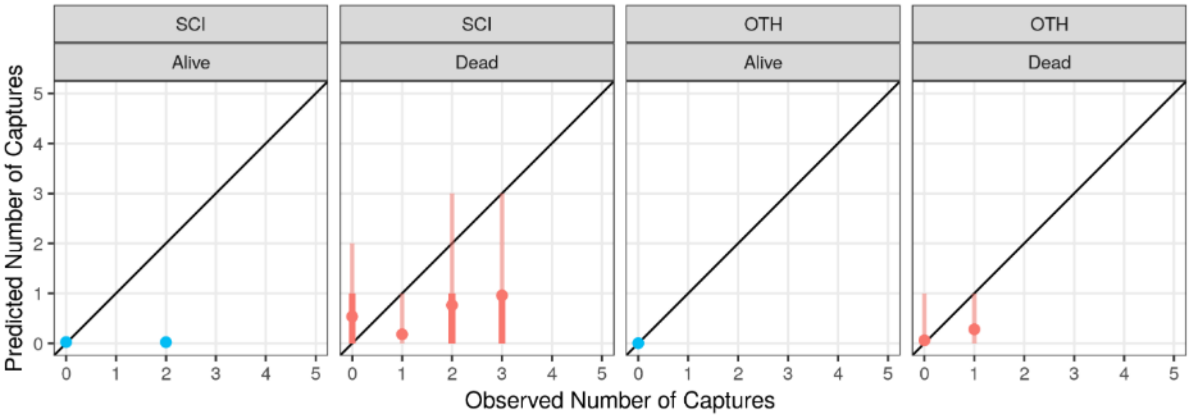

Figure A4-4: MCMC model run fits to annual observed captures by fishery group: "SCI" = scampi trawl, "OTH" = trawls targeting all other species around the Auckland Islands. Separate plots are shown for each fishery group and for live (blue bars) or dead (red bars) captures. Each plot shows annual observed captures estimated by the risk model (point = median, thick bar = interquartile range, whisker = 95% CI).

APPENDIX 5 SPATIAL RISK MODEL PARAMETER ESTIMATES

Table A5-1. Percentiles of estimated parameters and random variables from the spatial risk model for female New Zealand sea lions at the Auckland Islands.

		Quantiles of MCMC samples				
Parameter label	**Parameter description**	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
n_adults_t[1]	Female population size 1993	5 981	6 107	**6 694**	7 291	7 396
n_adults_t[2]	Female population size 1994	6 363	6 476	**7 042**	7 621	7 732
n_adults_t[3]	Female population size 1995	6 583	6 680	**7 172**	7 662	7 765
n_adults_t[4]	Female population size 1996	6 819	6 895	**7 335**	7 769	7 846
n_adults_t[5]	Female population size 1997	6 979	7 068	**7 445**	7 823	7 895
n_adults_t[6]	Female population size 1998	7 216	7 275	**7 606**	7 942	8 011
n_adults_t[7]	Female population size 1999	7 249	7 315	**7 639**	7 964	8 028
n_adults_t[8]	Female population size 2000	7 083	7 141	**7 433**	7 740	7 790
n_adults_t[9]	Female population size 2001	6 954	7 005	**7 270**	7 544	7 592
n_adults_t[10]	Female population size 2002	6 709	6 751	**7 003**	7 264	7 310
n_adults_t[11]	Female population size 2003	6 474	6 527	**6 774**	7 013	7 062
n_adults_t[12]	Female population size 2004	6 286	6 332	**6 570**	6 806	6 857
n_adults_t[13]	Female population size 2005	6 016	6 059	**6 297**	6 539	6 580
n_adults_t[14]	Female population size 2006	5 564	5 600	**5 826**	6 053	6 098
n_adults_t[15]	Female population size 2007	5 340	5 381	**5 596**	5 810	5 853
n_adults_t[16]	Female population size 2008	4 963	4 998	**5 187**	5 376	5 414
n_adults_t[17]	Female population size 2009	4 585	4 617	**4 781**	4 944	4 976
n_adults_t[18]	Female population size 2010	4 635	4 667	**4 828**	4 993	5 022
n_adults_t[19]	Female population size 2011	4 439	4 470	**4 634**	4 801	4 833
n_adults_t[20]	Female population size 2012	4 318	4 349	**4 509**	4 668	4 697
n_adults_t[21]	Female population size 2013	4 324	4 358	**4 532**	4 714	4 747
n_adults_t[22]	Female population size 2014	3 962	4 005	**4 234**	4 459	4 497
n_adults_t[23]	Female population size 2015	3 729	3 786	**4 104**	4 424	4 486
n_adults_t[24]	Female population size 2016	3 514	3 593	**4 083**	4 561	4 655
n_adults_t[25]	Female population size 2017	3 640	3 743	**4 326**	4 900	5 015
vulnerability_g[1]	Catchability squid bottom trawl no sled	0.044	0.048	**0.079**	0.122	0.131
vulnerability_g[2]	Catchability squid bottom trawl non-standard sled	0.057	0.062	**0.093**	0.133	0.142
vulnerability_g[3]	Catchability squid bottom trawl standard sled	0.016	0.018	**0.033**	0.054	0.058
vulnerability_g[4]	Catchability squid midwater trawl no sled	0.095	0.099	**0.126**	0.157	0.165
vulnerability_g[5]	Catchability squid midwater trawl non-standard sled	0.087	0.092	**0.118**	0.149	0.156
vulnerability_g[6]	Catchability squid midwater tawl standard sled	0.004	0.005	**0.017**	0.044	0.051
vulnerability_g[7]	Catchability scampi trawl	0.101	0.112	**0.186**	0.287	0.309
vulnerability_g[8]	Catchability other species trawl	0.032	0.050	**0.231**	0.649	0.760
p_observable[1]	Proportion captures observable squid bottom trawl no SLED	1.000	1.000	**1.000**	1.000	1.000
p_observable [2]	Proportion captures observable squid bottom trawl non-standard SLED	1.000	1.000	**1.000**	1.000	1.000
p_observable [3]	Proportion captures observable squid bottom trawl standard SLED	0.754	0.778	**0.874**	0.940	0.950
p_observable [4]	Proportion captures observable squid midwater trawl no SLED	1.000	1.000	**1.000**	1.000	1.000
p_observable [5]	Proportion captures observable squid midwater trawl non-standard SLED	1.000	1.000	**1.000**	1.000	1.000
p_observable [6]	Proportion captures observable squid midwater trawl standard SLED	0.415	0.453	**0.659**	0.827	0.854
p_observable [7]	Proportion captures observable scampi trawl	1.000	1.000	**1.000**	1.000	1.000
p_observable [8]	Proportion captures observable other species trawl	1.000	1.000	**1.000**	1.000	1.000
p_live_capture	Probability live capture	0.018	0.021	**0.042**	0.072	0.079
p_survive_capture	Probability survive release	0.509	0.519	**0.698**	0.881	0.892
rmax	r^{max}	0.100	0.103	**0.120**	0.137	0.140

APPENDIX 6 SPATIAL RISK MODEL ESTIMATES BY YEAR

Table A6-1. Quantiles of estimated annual deaths of female New Zealand sea lions for bottom trawls targeting southern arrow squid without a SLED.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0	0	**1**	2	3
1993/94	7	8	**17**	29	32
1994/95	6	8	**16**	28	30
1995/96	3	4	**11**	19	21
1996/97	7	9	**18**	30	33
1997/98	0	0	**3**	7	8
1998/99	0	0	**1**	3	3
1999/00	0	1	**4**	9	10
2000/01	0	0	**0**	1	1
2001/02	0	1	**3**	8	9
2002/03	0	0	**1**	3	3
2003/04	0	0	**0**	2	2
2004/05	0	0	**0**	2	2
2005/06	0	0	**0**	1	1
2006/07	0	0	**0**	0	0
2007/08	0	0	**0**	0	0
2008/09	0	0	**0**	0	0
2009/10	0	0	**0**	0	0
2010/11	0	0	**0**	0	0
2011/12	0	0	**0**	0	0
2012/13	0	0	**0**	0	0
2013/14	0	0	**0**	0	0
2014/15	0	0	**0**	0	0
2015/16	0	0	**0**	0	0
2016/17	0	0	**0**	0	0

Table A6-2. Quantiles of estimated annual deaths of female New Zealand sea lions for bottom trawls targeting southern arrow squid using a non-standardised SLED. Base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0	0	**0**	0	0
1993/94	0	0	**0**	0	0
1994/95	0	0	**0**	0	0
1995/96	0	0	**0**	0	0
1996/97	0	0	**0**	0	0
1997/98	0	0	**0**	0	0
1998/99	0	0	**0**	0	0
1999/00	0	0	**0**	0	0
2000/01	0	0	**2**	5	5
2001/02	0	0	**0**	2	2
2002/03	1	2	**6**	11	12
2003/04	4	5	**10**	18	19
2004/05	6	7	**15**	25	27
2005/06	12	14	**25**	39	42
2006/07	2	3	**7**	13	14
2007/08	2	2	**7**	12	14
2008/09	0	0	**0**	0	0
2009/10	0	0	**0**	0	0
2010/11	0	0	**0**	0	0
2011/12	0	0	**0**	0	0
2012/13	0	0	**0**	0	0
2013/14	0	0	**0**	0	0
2014/15	0	0	**0**	0	0
2015/16	0	0	**0**	0	0
2016/17	0	0	**0**	0	0

Table A6-3. Quantiles of estimated annual deaths of female New Zealand sea lions for bottom trawls targeting southern arrow squid using a standardised SLED. Base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0	0	**0**	0	0
1993/94	0	0	**0**	0	0
1994/95	0	0	**0**	0	0
1995/96	0	0	**0**	0	0
1996/97	0	0	**0**	0	0
1997/98	0	0	**0**	0	0
1998/99	0	0	**0**	0	0
1999/00	0	0	**0**	0	0
2000/01	0	0	**0**	0	0
2001/02	0	0	**0**	0	0
2002/03	0	0	**0**	0	0
2003/04	0	0	**0**	0	0
2004/05	0	0	**0**	0	0
2005/06	0	0	**0**	0	0
2006/07	0	0	**0**	0	0
2007/08	0	0	**0**	0	0
2008/09	0	1	**3**	8	8
2009/10	0	1	**3**	8	9
2010/11	0	1	**3**	8	9
2011/12	0	0	**2**	5	6
2012/13	0	0	**2**	5	6
2013/14	0	0	**1**	3	3
2014/15	0	0	**0**	2	3
2015/16	0	0	**2**	5	5
2016/17	0	0	**3**	7	8

Table A6-4. Quantiles of estimated annual deaths of female New Zealand sea lions for midwater trawls targeting southern arrow squid without a SLED. Base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	3	4	**9**	15	16
1993/94	43	45	**64**	87	92
1994/95	37	40	**57**	77	81
1995/96	49	52	**72**	95	101
1996/97	20	22	**33**	47	50
1997/98	14	16	**25**	35	38
1998/99	1	2	**6**	10	11
1999/00	7	8	**15**	22	24
2000/01	0	0	**0**	1	2
2001/02	0	0	**3**	6	7
2002/03	0	0	**2**	5	5
2003/04	0	0	**0**	2	2
2004/05	0	0	**0**	1	2
2005/06	0	0	**0**	2	2
2006/07	0	0	**0**	1	2
2007/08	0	0	**0**	1	1
2008/09	0	0	**0**	0	0
2009/10	0	0	**0**	0	0
2010/11	0	0	**0**	0	0
2011/12	0	0	**0**	0	0
2012/13	0	0	**0**	0	0
2013/14	0	0	**0**	0	0
2014/15	0	0	**0**	0	0
2015/16	0	0	**0**	0	0
2016/17	0	0	**0**	0	0

Table A6-5. Quantiles of estimated annual deaths of female New Zealand sea lions for midwater trawls targeting southern arrow squid using a non-standardised SLED. Base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0	0	**0**	0	0
1993/94	0	0	**0**	0	0
1994/95	0	0	**0**	0	0
1995/96	0	0	**0**	0	0
1996/97	0	0	**0**	0	0
1997/98	0	0	**0**	0	0
1998/99	0	0	**0**	0	0
1999/00	0	0	**0**	0	0
2000/01	2	2	**6**	11	12
2001/02	8	9	**16**	24	26
2002/03	6	7	**14**	21	23
2003/04	11	12	**20**	29	32
2004/05	11	13	**21**	31	34
2005/06	3	4	**9**	15	16
2006/07	3	4	**9**	15	16
2007/08	4	5	**10**	16	18
2008/09	0	0	**0**	0	0
2009/10	0	0	**0**	0	0
2010/11	0	0	**0**	0	0
2011/12	0	0	**0**	0	0
2012/13	0	0	**0**	0	0
2013/14	0	0	**0**	0	0
2014/15	0	0	**0**	0	0
2015/16	0	0	**0**	0	0
2016/17	0	0	**0**	0	0

Table A6-6. Quantiles of estimated annual deaths of female New Zealand sea lions for midwater trawls targeting southern arrow squid using a standardised SLED. Base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0	0	**0**	0	0
1993/94	0	0	**0**	0	0
1994/95	0	0	**0**	0	0
1995/96	0	0	**0**	0	0
1996/97	0	0	**0**	0	0
1997/98	0	0	**0**	0	0
1998/99	0	0	**0**	0	0
1999/00	0	0	**0**	0	0
2000/01	0	0	**0**	0	0
2001/02	0	0	**0**	0	0
2002/03	0	0	**0**	0	0
2003/04	0	0	**0**	0	0
2004/05	0	0	**0**	0	0
2005/06	0	0	**0**	0	0
2006/07	0	0	**0**	0	0
2007/08	0	0	**0**	0	0
2008/09	0	0	**2**	6	7
2009/10	0	0	**0**	2	2
2010/11	0	0	**0**	2	3
2011/12	0	0	**0**	2	3
2012/13	0	0	**0**	2	2
2013/14	0	0	**0**	2	2
2014/15	0	0	**0**	2	3
2015/16	0	0	**1**	3	4
2016/17	0	0	**0**	2	2

Table A6-7. Quantiles of estimated annual deaths of female New Zealand sea lions for trawls targeting scampi. Base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	1	1	**5**	10	12
1993/94	4	5	**12**	21	23
1994/95	3	4	**9**	17	19
1995/96	2	3	**8**	16	17
1996/97	1	2	**6**	12	13
1997/98	2	2	**7**	13	15
1998/99	2	3	**8**	15	16
1999/00	1	2	**6**	11	12
2000/01	1	2	**6**	12	13
2001/02	1	2	**6**	12	13
2002/03	1	1	**5**	10	11
2003/04	2	2	**7**	13	15
2004/05	2	3	**8**	15	17
2005/06	1	2	**6**	12	14
2006/07	1	2	**6**	12	13
2007/08	1	1	**5**	10	11
2008/09	2	2	**7**	13	14
2009/10	0	0	**2**	6	7
2010/11	1	2	**5**	10	12
2011/12	0	1	**3**	8	9
2012/13	0	0	**3**	7	8
2013/14	0	0	**2**	6	7
2014/15	0	0	**1**	4	4
2015/16	0	1	**3**	8	8
2016/17	1	1	**5**	9	11

Table A6-8. Quantiles of estimated annual deaths of female New Zealand sea lions for trawls targeting all other species. Base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0	0	**0**	2	2
1993/94	0	0	**1**	4	5
1994/95	0	0	**1**	3	4
1995/96	0	0	**1**	6	7
1996/97	0	0	**1**	5	6
1997/98	0	0	**2**	8	10
1998/99	0	0	**1**	4	5
1999/00	0	0	**1**	5	6
2000/01	0	0	**0**	3	3
2001/02	0	0	**1**	5	6
2002/03	0	0	**1**	5	6
2003/04	0	0	**0**	1	2
2004/05	0	0	**0**	1	2
2005/06	0	0	**0**	1	1
2006/07	0	0	**0**	0	1
2007/08	0	0	**0**	1	1
2008/09	0	0	**0**	1	1
2009/10	0	0	**0**	1	1
2010/11	0	0	**0**	1	2
2011/12	0	0	**0**	1	1
2012/13	0	0	**0**	1	1
2013/14	0	0	**0**	1	2
2014/15	0	0	**0**	2	2
2015/16	0	0	**0**	1	2
2016/17	0	0	**0**	2	2

Table A6-9. Quantiles of estimated annual deaths of female New Zealand sea lions across all Auckland Islands trawl fishery groups. Base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	8	9	**15**	24	25
1993/94	68	72	**96**	123	129
1994/95	60	63	**85**	109	114
1995/96	67	71	**94**	120	126
1996/97	40	43	**60**	79	83
1997/98	24	26	**38**	52	55
1998/99	8	9	**16**	25	27
1999/00	16	17	**26**	37	39
2000/01	8	9	**15**	23	25
2001/02	19	21	**31**	42	45
2002/03	18	20	**29**	40	42
2003/04	25	27	**39**	53	55
2004/05	31	33	**46**	61	65
2005/06	26	28	**41**	57	60
2006/07	13	14	**23**	33	35
2007/08	12	14	**22**	32	34
2008/09	5	6	**12**	21	23
2009/10	2	2	**6**	12	13
2010/11	4	4	**10**	17	18
2011/12	2	2	**6**	12	13
2012/13	2	2	**6**	11	12
2013/14	1	1	**4**	8	9
2014/15	0	0	**3**	7	7
2015/16	2	2	**6**	12	13
2016/17	3	4	**8**	15	16

Table A6-10. Quantiles of estimated annual population sustainability threshold (PST) of female New Zealand sea lions at the Auckland Islands. Produced by the base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	32.40	33.65	**40.15**	47.09	48.43
1993/94	34.18	35.48	**42.25**	49.19	50.62
1994/95	35.35	36.43	**43.04**	49.70	50.90
1995/96	36.24	37.39	**43.95**	50.78	52.04
1996/97	36.94	38.10	**44.66**	51.24	52.43
1997/98	37.73	38.90	**45.63**	52.40	53.76
1998/99	37.76	39.14	**45.87**	52.55	53.92
1999/00	36.98	38.18	**44.59**	51.17	52.35
2000/01	36.21	37.31	**43.62**	49.96	51.10
2001/02	34.85	36.00	**42.07**	48.04	49.13
2002/03	33.83	34.86	**40.69**	46.48	47.56
2003/04	32.69	33.78	**39.38**	45.09	46.27
2004/05	31.35	32.36	**37.76**	43.27	44.32
2005/06	29.06	29.96	**34.99**	40.00	40.97
2006/07	27.76	28.76	**33.61**	38.37	39.38
2007/08	25.76	26.72	**31.16**	35.55	36.56
2008/09	23.77	24.60	**28.68**	32.81	33.62
2009/10	24.04	24.87	**28.99**	33.11	33.89
2010/11	23.07	23.85	**27.81**	31.82	32.57
2011/12	22.42	23.17	**27.04**	31.05	31.70
2012/13	22.51	23.31	**27.21**	31.15	31.86
2013/14	20.88	21.68	**25.39**	29.25	29.96
2014/15	20.00	20.73	**24.57**	28.67	29.34
2015/16	19.34	20.15	**24.40**	29.02	29.99
2016/17	20.19	21.06	**25.96**	31.12	32.15

Table A6-11. Quantiles of estimated annual risk ratio of female New Zealand sea lions for bottom trawls targeting southern arrow squid without a SLED. Produced by the base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0.000	0.000	**0.021**	0.059	0.074
1993/94	0.165	0.195	**0.403**	0.701	0.774
1994/95	0.149	0.179	**0.377**	0.658	0.725
1995/96	0.077	0.099	**0.242**	0.448	0.495
1996/97	0.156	0.187	**0.398**	0.697	0.765
1997/98	0.000	0.000	**0.064**	0.151	0.17
1998/99	0.000	0.000	**0.020**	0.062	0.07
1999/00	0.000	0.022	**0.090**	0.199	0.225
2000/01	0.000	0.000	**0.000**	0.024	0.026
2001/02	0.000	0.021	**0.080**	0.185	0.209
2002/03	0.000	0.000	**0.023**	0.071	0.08
2003/04	0.000	0.000	**0.000**	0.053	0.061
2004/05	0.000	0.000	**0.000**	0.053	0.057
2005/06	0.000	0.000	**0.000**	0.031	0.035
2006/07	0.000	0.000	**0.000**	0.000	0.000
2007/08	0.000	0.000	**0.000**	0.000	0.000
2008/09	0.000	0.000	**0.000**	0.000	0.000
2009/10	0.000	0.000	**0.000**	0.000	0.000
2010/11	0.000	0.000	**0.000**	0.000	0.000
2011/12	0.000	0.000	**0.000**	0.000	0.000
2012/13	0.000	0.000	**0.000**	0.000	0.000
2013/14	0.000	0.000	**0.000**	0.000	0.000
2014/15	0.000	0.000	**0.000**	0.000	0.000
2015/16	0.000	0.000	**0.000**	0.000	0.000
2016/17	0.000	0.000	**0.000**	0.000	0.000

Table A6-12. Quantiles of estimated annual risk ratio of female New Zealand sea lions for bottom trawls targeting southern arrow squid using a non-standardised SLED. Produced by the base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0.000	0.000	**0.000**	0.000	0.000
1993/94	0.000	0.000	**0.000**	0.000	0.000
1994/95	0.000	0.000	**0.000**	0.000	0.000
1995/96	0.000	0.000	**0.000**	0.000	0.000
1996/97	0.000	0.000	**0.000**	0.000	0.000
1997/98	0.000	0.000	**0.000**	0.000	0.000
1998/99	0.000	0.000	**0.000**	0.000	0.000
1999/00	0.000	0.000	**0.000**	0.000	0.000
2000/01	0.000	0.000	**0.045**	0.114	0.131
2001/02	0.000	0.000	**0.000**	0.047	0.052
2002/03	0.026	0.047	**0.136**	0.265	0.296
2003/04	0.091	0.114	**0.260**	0.458	0.508
2004/05	0.161	0.191	**0.394**	0.672	0.733
2005/06	0.34	0.389	**0.706**	1.142	1.242
2006/07	0.059	0.081	**0.208**	0.394	0.439
2007/08	0.061	0.074	**0.209**	0.404	0.45
2008/09	0.000	0.000	**0.000**	0.000	0.000
2009/10	0.000	0.000	**0.000**	0.000	0.000
2010/11	0.000	0.000	**0.000**	0.000	0.000
2011/12	0.000	0.000	**0.000**	0.000	0.000
2012/13	0.000	0.000	**0.000**	0.000	0.000
2013/14	0.000	0.000	**0.000**	0.000	0.000
2014/15	0.000	0.000	**0.000**	0.000	0.000
2015/16	0.000	0.000	**0.000**	0.000	0.000
2016/17	0.000	0.000	**0.000**	0.000	0.000

Table A6-13. Quantiles of estimated annual risk ratio of female New Zealand sea lions for bottom trawls targeting southern arrow squid using a standardised SLED. Produced by the base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0.000	0.000	**0.000**	0.000	0.000
1993/94	0.000	0.000	**0.000**	0.000	0.000
1994/95	0.000	0.000	**0.000**	0.000	0.000
1995/96	0.000	0.000	**0.000**	0.000	0.000
1996/97	0.000	0.000	**0.000**	0.000	0.000
1997/98	0.000	0.000	**0.000**	0.000	0.000
1998/99	0.000	0.000	**0.000**	0.000	0.000
1999/00	0.000	0.000	**0.000**	0.000	0.000
2000/01	0.000	0.000	**0.000**	0.000	0.000
2001/02	0.000	0.000	**0.000**	0.000	0.000
2002/03	0.000	0.000	**0.000**	0.000	0.000
2003/04	0.000	0.000	**0.000**	0.000	0.000
2004/05	0.000	0.000	**0.000**	0.000	0.000
2005/06	0.000	0.000	**0.000**	0.000	0.000
2006/07	0.000	0.000	**0.000**	0.000	0.000
2007/08	0.000	0.000	**0.000**	0.000	0.000
2008/09	0.000	0.030	**0.111**	0.271	0.301
2009/10	0.000	0.030	**0.111**	0.268	0.308
2010/11	0.000	0.031	**0.122**	0.285	0.326
2011/12	0.000	0.000	**0.075**	0.196	0.233
2012/13	0.000	0.000	**0.075**	0.206	0.234
2013/14	0.000	0.000	**0.037**	0.118	0.136
2014/15	0.000	0.000	**0.000**	0.096	0.124
2015/16	0.000	0.000	**0.072**	0.192	0.219
2016/17	0.000	0.000	**0.107**	0.263	0.301

Table A6-14. Quantiles of estimated annual risk ratio of female New Zealand sea lions for midwater trawls targeting southern arrow squid without a SLED. Produced by the base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0.079	0.1	**0.221**	0.382	0.418
1993/94	0.991	1.063	**1.531**	2.121	2.236
1994/95	0.841	0.914	**1.332**	1.843	1.969
1995/96	1.081	1.156	**1.64**	2.257	2.4
1996/97	0.441	0.48	**0.746**	1.086	1.15
1997/98	0.308	0.341	**0.544**	0.802	0.856
1998/99	0.024	0.042	**0.121**	0.229	0.254
1999/00	0.157	0.181	**0.328**	0.514	0.557
2000/01	0.000	0.000	**0.000**	0.027	0.044
2001/02	0.000	0.000	**0.063**	0.139	0.159
2002/03	0.000	0.000	**0.048**	0.12	0.137
2003/04	0.000	0.000	**0.000**	0.052	0.058
2004/05	0.000	0.000	**0.000**	0.03	0.048
2005/06	0.000	0.000	**0.000**	0.057	0.064
2006/07	0.000	0.000	**0.000**	0.034	0.058
2007/08	0.000	0.000	**0.000**	0.033	0.035
2008/09	0.000	0.000	**0.000**	0.000	0.000
2009/10	0.000	0.000	**0.000**	0.000	0.000
2010/11	0.000	0.000	**0.000**	0.000	0.000
2011/12	0.000	0.000	**0.000**	0.000	0.000
2012/13	0.000	0.000	**0.000**	0.000	0.000
2013/14	0.000	0.000	**0.000**	0.000	0.000
2014/15	0.000	0.000	**0.000**	0.000	0.000
2015/16	0.000	0.000	**0.000**	0.000	0.000
2016/17	0.000	0.000	**0.000**	0.000	0.000

Table A6-15. Quantiles of estimated annual risk ratio of female New Zealand sea lions for midwater trawls targeting southern arrow squid using a non-standardised SLED. Produced by the base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0.000	0.000	**0.000**	0.000	0.000
1993/94	0.000	0.000	**0.000**	0.000	0.000
1994/95	0.000	0.000	**0.000**	0.000	0.000
1995/96	0.000	0.000	**0.000**	0.000	0.000
1996/97	0.000	0.000	**0.000**	0.000	0.000
1997/98	0.000	0.000	**0.000**	0.000	0.000
1998/99	0.000	0.000	**0.000**	0.000	0.000
1999/00	0.000	0.000	**0.000**	0.000	0.000
2000/01	0.040	0.047	**0.134**	0.250	0.278
2001/02	0.186	0.212	**0.375**	0.588	0.632
2002/03	0.155	0.178	**0.333**	0.532	0.577
2003/04	0.267	0.298	**0.504**	0.770	0.822
2004/05	0.297	0.335	**0.558**	0.848	0.918
2005/06	0.095	0.117	**0.259**	0.442	0.482
2006/07	0.098	0.125	**0.268**	0.459	0.510
2007/08	0.129	0.156	**0.323**	0.544	0.588
2008/09	0.000	0.000	**0.000**	0.000	0.000
2009/10	0.000	0.000	**0.000**	0.000	0.000
2010/11	0.000	0.000	**0.000**	0.000	0.000
2011/12	0.000	0.000	**0.000**	0.000	0.000
2012/13	0.000	0.000	**0.000**	0.000	0.000
2013/14	0.000	0.000	**0.000**	0.000	0.000
2014/15	0.000	0.000	**0.000**	0.000	0.000
2015/16	0.000	0.000	**0.000**	0.000	0.000
2016/17	0.000	0.000	**0.000**	0.000	0.000

Table A6-16. Quantiles of estimated annual risk ratio of female New Zealand sea lions for midwater trawls targeting southern arrow squid using a standardised SLED. Produced by the base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0.000	0.000	**0.000**	0.000	0.000
1993/94	0.000	0.000	**0.000**	0.000	0.000
1994/95	0.000	0.000	**0.000**	0.000	0.000
1995/96	0.000	0.000	**0.000**	0.000	0.000
1996/97	0.000	0.000	**0.000**	0.000	0.000
1997/98	0.000	0.000	**0.000**	0.000	0.000
1998/99	0.000	0.000	**0.000**	0.000	0.000
1999/00	0.000	0.000	**0.000**	0.000	0.000
2000/01	0.000	0.000	**0.000**	0.000	0.000
2001/02	0.000	0.000	**0.000**	0.000	0.000
2002/03	0.000	0.000	**0.000**	0.000	0.000
2003/04	0.000	0.000	**0.000**	0.000	0.000
2004/05	0.000	0.000	**0.000**	0.000	0.000
2005/06	0.000	0.000	**0.000**	0.000	0.000
2006/07	0.000	0.000	**0.000**	0.000	0.000
2007/08	0.000	0.000	**0.000**	0.000	0.000
2008/09	0.000	0.000	**0.066**	0.211	0.258
2009/10	0.000	0.000	**0.000**	0.071	0.081
2010/11	0.000	0.000	**0.000**	0.080	0.106
2011/12	0.000	0.000	**0.000**	0.081	0.108
2012/13	0.000	0.000	**0.000**	0.072	0.080
2013/14	0.000	0.000	**0.000**	0.079	0.091
2014/15	0.000	0.000	**0.000**	0.091	0.121
2015/16	0.000	0.000	**0.036**	0.127	0.159
2016/17	0.000	0.000	**0.000**	0.080	0.096

Table A6-17. Quantiles of estimated annual risk ratio of female New Zealand sea lions for scampi trawls. Produced by the base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0.024	0.028	**0.128**	0.262	0.297
1993/94	0.101	0.126	**0.283**	0.517	0.570
1994/95	0.070	0.089	**0.219**	0.402	0.448
1995/96	0.054	0.071	**0.191**	0.366	0.405
1996/97	0.023	0.042	**0.134**	0.266	0.298
1997/98	0.041	0.049	**0.151**	0.294	0.329
1998/99	0.046	0.063	**0.172**	0.334	0.368
1999/00	0.023	0.042	**0.127**	0.256	0.288
2000/01	0.024	0.044	**0.134**	0.271	0.309
2001/02	0.026	0.046	**0.144**	0.286	0.317
2002/03	0.023	0.026	**0.118**	0.250	0.278
2003/04	0.046	0.057	**0.178**	0.350	0.389
2004/05	0.061	0.081	**0.218**	0.417	0.461
2005/06	0.033	0.057	**0.175**	0.356	0.393
2006/07	0.031	0.057	**0.176**	0.355	0.398
2007/08	0.030	0.033	**0.153**	0.322	0.362
2008/09	0.061	0.071	**0.232**	0.463	0.514
2009/10	0.000	0.000	**0.077**	0.206	0.239
2010/11	0.035	0.059	**0.186**	0.381	0.427
2011/12	0.000	0.034	**0.126**	0.285	0.328
2012/13	0.000	0.000	**0.108**	0.255	0.287
2013/14	0.000	0.000	**0.088**	0.239	0.270
2014/15	0.000	0.000	**0.042**	0.154	0.176
2015/16	0.000	0.036	**0.136**	0.315	0.350
2016/17	0.036	0.041	**0.175**	0.369	0.414

Table A6-18. Quantiles of estimated annual risk ratio of female New Zealand sea lions for trawls targeting all other species. Produced by the base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0.000	0.000	**0.000**	0.046	0.053
1993/94	0.000	0.000	**0.023**	0.094	0.116
1994/95	0.000	0.000	**0.020**	0.073	0.090
1995/96	0.000	0.000	**0.026**	0.129	0.160
1996/97	0.000	0.000	**0.025**	0.122	0.143
1997/98	0.000	0.000	**0.048**	0.182	0.220
1998/99	0.000	0.000	**0.021**	0.089	0.106
1999/00	0.000	0.000	**0.023**	0.105	0.131
2000/01	0.000	0.000	**0.000**	0.068	0.078
2001/02	0.000	0.000	**0.025**	0.117	0.143
2002/03	0.000	0.000	**0.026**	0.125	0.150
2003/04	0.000	0.000	**0.000**	0.031	0.051
2004/05	0.000	0.000	**0.000**	0.033	0.054
2005/06	0.000	0.000	**0.000**	0.030	0.032
2006/07	0.000	0.000	**0.000**	0.000	0.029
2007/08	0.000	0.000	**0.000**	0.033	0.035
2008/09	0.000	0.000	**0.000**	0.036	0.039
2009/10	0.000	0.000	**0.000**	0.035	0.038
2010/11	0.000	0.000	**0.000**	0.041	0.070
2011/12	0.000	0.000	**0.000**	0.038	0.042
2012/13	0.000	0.000	**0.000**	0.039	0.044
2013/14	0.000	0.000	**0.000**	0.048	0.080
2014/15	0.000	0.000	**0.000**	0.078	0.089
2015/16	0.000	0.000	**0.000**	0.047	0.075
2016/17	0.000	0.000	**0.000**	0.070	0.081

Table A6-19. Quantiles of estimated annual female New Zealand sea lion risk ratio across all Auckland Islands trawl fisheries. Produced by the base case model run, using a calibration coefficient (ϕ) of 0.1.

	Quantiles of MCMC samples				
Fishing year	**2.50%**	**5.00%**	**50.0%**	**95.00%**	**97.50%**
1992/93	0.19	0.21	**0.38**	0.60	0.65
1993/94	1.58	1.68	**2.27**	3.00	3.16
1994/95	1.35	1.44	**1.97**	2.63	2.76
1995/96	1.49	1.58	**2.14**	2.84	3.00
1996/97	0.87	0.94	**1.34**	1.82	1.94
1997/98	0.52	0.56	**0.83**	1.17	1.25
1998/99	0.17	0.19	**0.34**	0.55	0.60
1999/00	0.35	0.38	**0.59**	0.86	0.92
2000/01	0.18	0.20	**0.35**	0.55	0.60
2001/02	0.45	0.49	**0.73**	1.03	1.11
2002/03	0.44	0.47	**0.71**	1.02	1.09
2003/04	0.63	0.68	**0.99**	1.38	1.46
2004/05	0.79	0.84	**1.22**	1.68	1.77
2005/06	0.73	0.79	**1.18**	1.69	1.80
2006/07	0.38	0.42	**0.68**	1.01	1.08
2007/08	0.39	0.44	**0.71**	1.07	1.15
2008/09	0.18	0.21	**0.44**	0.74	0.81
2009/10	0.06	0.08	**0.22**	0.42	0.47
2010/11	0.13	0.16	**0.35**	0.61	0.67
2011/12	0.07	0.08	**0.24**	0.44	0.50
2012/13	0.06	0.07	**0.21**	0.41	0.46
2013/14	0.03	0.04	**0.16**	0.33	0.37
2014/15	0.00	0.00	**0.12**	0.27	0.30
2015/16	0.08	0.09	**0.26**	0.49	0.54
2016/17	0.11	0.14	**0.32**	0.58	0.65

APPENDIX 7 SPATIAL RISK MODEL CODE

```
/**
 * Sub-Antarctic sea lion (HSL) spatially explicit fisheries risk assessment (SEFRA) model
 *
 * Darcy Webber
 * darcy@quantifish.co.nz
 *
 * Kath Large
 * NIWA
 *
 * July 2019
**/

data {

  // Dimensions
  int<lower=1> n_species;
  int<lower=1> n_method;
  int<lower=n_method> n_fishery_group;
  int<lower=1> n_years;

  // Observed fishing events and observed captures
  int<lower=1> n_i;
  int<lower=1,upper=n_fishery_group> fishery_group_i[n_i];
  vector<lower=0>[n_i] overlap_i;
  int<lower=0> live_captures_i[n_i];
  int<lower=0> dead_captures_i[n_i];
  int<lower=1,upper=n_years> years_i[n_i];

  // All fishing events (this includes observed and unobserved events)
  int<lower=1> n_j;
  int<lower=1,upper=n_fishery_group> fishery_group_j[n_j];
  vector<lower=0>[n_j] overlap_j;
  int<lower=1,upper=n_years> years_j[n_j];

  // Priors for demographic parameters
  vector[n_years] mu_n_t;
  vector<lower=0>[n_years] sd_n_t;
  real mu_rmax;
  real<lower=0> sd_rmax;

  // PST calculation management target
  real<lower=0,upper=1> psi;

} // end of data

parameters {

  vector[8] log_v_g;
  real<lower=0,upper=1> p_live_capture;
  vector<lower=0,upper=1>[n_fishery_group] p_observable_g;
  vector<lower=0>[n_years] n_adults_t;

} // end of parameters

transformed parameters {

  vector[n_fishery_group] vulnerability_g;
  vector[n_i] mu_live_captures_i;
  vector[n_i] mu_dead_captures_i;

  for (g in 1:n_fishery_group) {
    vulnerability_g[g] = exp(log_v_g[g]);
  }

  for (i in 1:n_i) {
    real mu_captures = vulnerability_g[fishery_group_i[i]] *
p_observable_g[fishery_group_i[i]] * overlap_i[i] * n_adults_t[years_i[i]];
    mu_live_captures_i[i] = mu_captures * p_live_capture;
    mu_dead_captures_i[i] = mu_captures * (1.0 - p_live_capture);
  }

} // end of transformed parameters
```

```
model {

  // Prior
  log_v_g ~ normal(0.0, 10.0);
  p_live_capture ~ beta(1.0, 3.0);
  p_observable_g[1] ~ beta(300,0.001);
  p_observable_g[2] ~ beta(300,0.001);
  p_observable_g[3] ~ beta(39.224658,5.885302);
  p_observable_g[4] ~ beta(300,0.001);
  p_observable_g[5] ~ beta(300,0.001);
  p_observable_g[6] ~ beta(10.617046,5.669724);
  p_observable_g[7] ~ beta(300,0.001);
  p_observable_g[8] ~ beta(300,0.001);

  for (t in 1:n_years) {
    n_adults_t[t] ~ normal(mu_n_t[t], sd_n_t[t]);
  }

  // Likelihood
  live_captures_i ~ poisson(mu_live_captures_i);
  dead_captures_i ~ poisson(mu_dead_captures_i);

} // end of model

generated quantities {

  // Outputs for observed and unobserved effort
  vector[n_j] mu_captures_j;
  vector[n_j] mu_deaths_j;
  vector[n_j] mu_risk_j;
  vector[n_j] captures_j;
  vector[n_j] deaths_j;

  // Prior and Posterior predictive checking
  vector[n_fishery_group] prior_log_v_g;
  real prior_p_live_capture;
  real prior_p_survive_capture;
  vector[n_fishery_group] prior_p_observable_g;
  vector[n_years] prior_n_adults_t;
  vector[n_i] pred_live_captures_i;
  vector[n_i] pred_dead_captures_i;

  // Captures
  real captures;
  vector[n_years] captures_t;
  vector[n_fishery_group] captures_g_3y;
  matrix[n_years,n_fishery_group] captures_tg;

  // Deaths
  real deaths;
  vector[n_years] deaths_t;
  vector[n_fishery_group] deaths_g_3y;
  matrix[n_years,n_fishery_group] deaths_tg;

  // rmax, PST, risk ratio
  vector<lower=0,upper=1>[n_method] p_survive_capture;
  vector<lower=0,upper=1>[n_species] rmax_s;
  vector[n_years] pst_t;
  real risk_ratio_3y;
  real risk_ratio_B_3y;
  vector[n_years] risk_ratio_t;
  vector[n_fishery_group] risk_ratio_gB_3y;
  matrix[n_years,n_fishery_group] risk_ratio_tg;

  // Prior checking and simulated parameters
  for (g in 1:n_fishery_group) {
    prior_log_v_g[g] = normal_rng(0.0, 10.0);
  }
  prior_p_live_capture = beta_rng(1.0, 3.0);
  prior_p_survive_capture = uniform_rng(0.5, 0.9);
  p_survive_capture[1] = prior_p_survive_capture;
  prior_p_observable_g[1] = beta_rng(100, 0.001);
  prior_p_observable_g[2] = beta_rng(100, 0.001);
  prior_p_observable_g[3] = beta_rng(39.224658,5.885302);
  prior_p_observable_g[4] = beta_rng(100, 0.001);
  prior_p_observable_g[5] = beta_rng(100, 0.001);
  prior_p_observable_g[6] = beta_rng(10.617046,5.669724);
```

```
  prior_p_observable_g[7] = beta_rng(100, 0.001);
  prior_p_observable_g[8] = beta_rng(100, 0.001);

  rmax_s[1] = normal_rng(mu_rmax, sd_rmax);
  for (t in 1:n_years) {
    prior_n_adults_t[t] = normal_rng(mu_n_t[t], sd_n_t[t]);
    pst_t[t] = 0.5 * psi * rmax_s[1] * n_adults_t[t];
  }

  // Posterior predictive checking
  for (i in 1:n_i) {
    pred_live_captures_i[i] = poisson_rng(mu_live_captures_i[i]);
    pred_dead_captures_i[i] = poisson_rng(mu_dead_captures_i[i]);
  }

  // Captures and deaths prediction
  for (g in 1:n_fishery_group) {
    for (t in 1:n_years) {
      captures_tg[t,g] = 0.0;
      deaths_tg[t,g] = 0.0;
    }
  }

  for (j in 1:n_j) {
    int gj = fishery_group_j[j];
    int tj = years_j[j];

    mu_captures_j[j] = vulnerability_g[gj] * p_observable_g[gj] * overlap_j[j] *
n_adults_t[tj];
    mu_deaths_j[j] = vulnerability_g[gj] * (1.0 - p_live_capture * prior_p_survive_capture) *
overlap_j[j] * n_adults_t[tj];
    mu_risk_j[j] = mu_deaths_j[j] / pst_t[tj];
    captures_j[j] = poisson_rng(mu_captures_j[j]);
    deaths_j[j] = poisson_rng(mu_deaths_j[j]);
    captures_tg[tj,gj] += captures_j[j];
    deaths_tg[tj,gj] += deaths_j[j];
  }

  // Outputs as matrices by species and fishery group for Risk Atlas
  /*
  for (g in 1:n_fishery_group) {
    vulnerability_tg[1,g] = vulnerability_g[g];
    p_observable_tg[1,g] = p_observable_g[g];
    p_live_capture_tg[1,g] = p_live_capture;
    p_survive_capture_tg[1,g] = prior_p_survive_capture;
  }
  */

  // PST and risk ratio calculations by year and fishery group
  for (t in 1:n_years) {
    captures_t[t] = sum(captures_tg[t]);
    deaths_t[t] = sum(deaths_tg[t]);
    risk_ratio_t[t] = deaths_t[t] / pst_t[t];
    for (g in 1:n_fishery_group) {
      risk_ratio_tg[t,g] = deaths_tg[t,g] / pst_t[t];
    }
  }
  captures = sum(captures_t);
  deaths = sum(deaths_t);
  risk_ratio_3y = (risk_ratio_t[23] + risk_ratio_t[24] + risk_ratio_t[25]) / 3.00;


//annual captures and deaths for last 3 years (2015 to 2017)
for (g in 1:n_fishery_group) {
    captures_g_3y[g] = (captures_tg[23,g] + captures_tg[24,g] + captures_tg[25,g]) / 3.00;
    deaths_g_3y[g] = (deaths_tg[23,g] + deaths_tg[24,g] + deaths_tg[25,g]) / 3.00;
}

//annual risk ratio by fishery group (based on last 3 years, 2015 to 2017)
for (g in 1:n_fishery_group) {
  risk_ratio_gB_3y[g] = (risk_ratio_tg[23,g] + risk_ratio_tg[24,g] + risk_ratio_tg[25,g]) /
3.00;
}
risk_ratio_B_3y = sum(risk_ratio_gB_3y);


} // end of generated quantities
```