# AEBAR 2019-20: *Snapshot* of Chapter 4 - New Zealand sea lion (*Phocarctos hookeri*)

**New Zealand sea lion**
(*Phocarctos hookeri*)

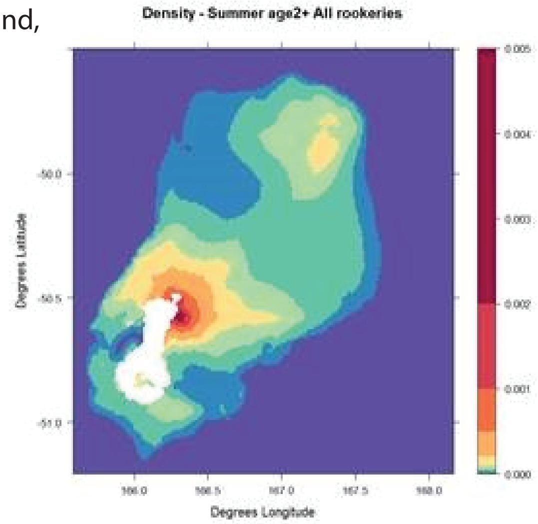
**Threatened -**
**Nationally vulnerable (DOC 2019)**

## 1. THE ISSUE IN BRIEF

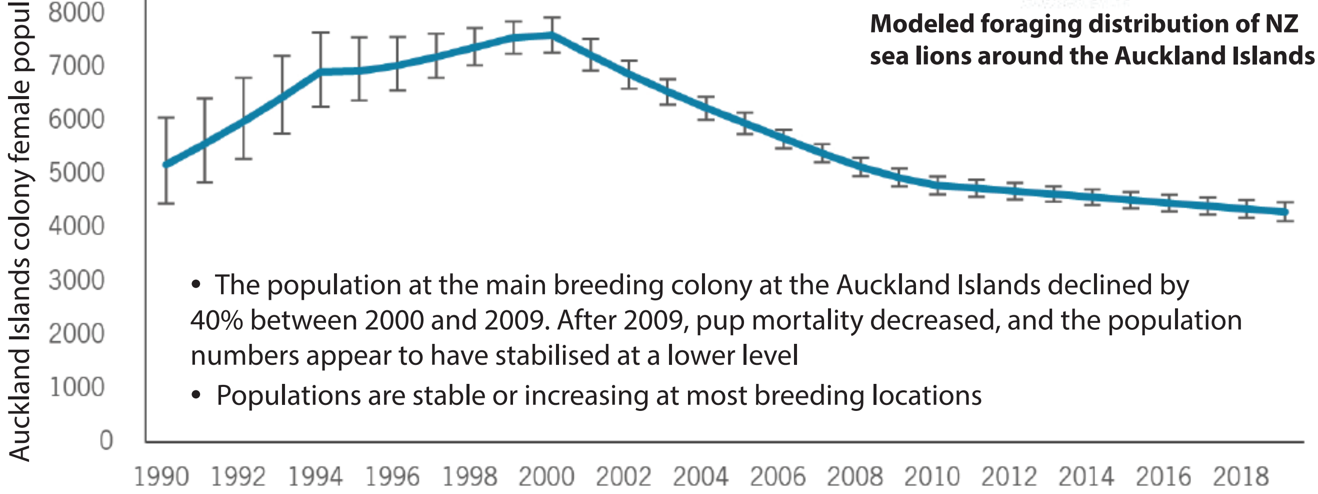
- The New Zealand sea lion (*Phocarctos hookeri*) is a pinniped, breeding only in New Zealand, classed as 'Nationally vulnerable' by the Department of Conservation
- The population of the main breeding colony at the Auckland Islands has declined from a peak in 2000
- Like all marine mammals, NZ sea lions are protected under the Marine Mammals Protection Act 1978 and the Fisheries Act 1996. NZ sea lions are managed under a Threat Management Plan (2017–2022)
- Potential threats to this species include human disturbance (on the mainland), direct and indirect effects of fisheries (for adults and sub-adults, see boxes 4 and 5), diseases, and possible climate effects

## 2. DISTRIBUTION AND ABUNDANCE

- New Zealand sea lions were once present throughout New Zealand, primarily in the southern regions, prior to human settlement
- Currently there are three recognised breeding colonies, on the Auckland Islands, Campbell Island, and Stewart Island, and recently established breeding sites on the Southland coast
- Population estimates are based on demographic models informed by annual pup counts and mark-recapture data
- Sea lions can roam up to 200 km away from the colony during foraging trips



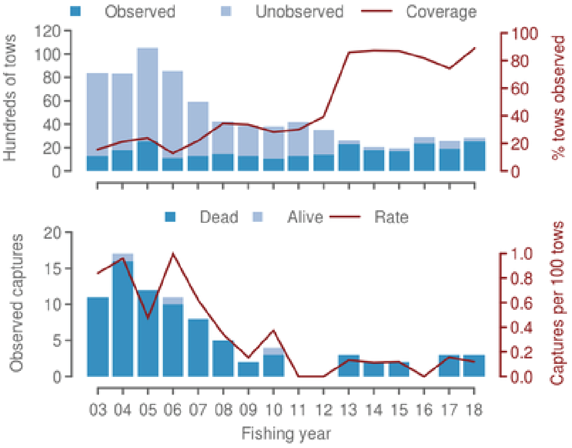
**Modeled foraging distribution of NZ sea lions around the Auckland Islands**

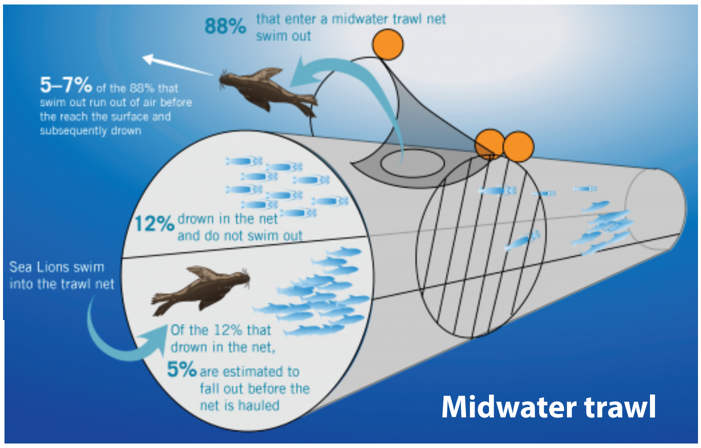


- The population at the main breeding colony at the Auckland Islands declined by 40% between 2000 and 2009. After 2009, pup mortality decreased, and the population numbers appear to have stabilised at a lower level
- Populations are stable or increasing at most breeding locations

# 3. FISHERIES INTERACTIONS

- Sea lions can enter trawl nets during fishing operations, and may drown in the net
- Sea Lion Exclusion Devices (SLEDs) enable sea lions to exit the net, reducing the risk of drowning. They were developed, and ultimately fully adopted from 2008. SLEDs are used in trawl fisheries near the Auckland Islands and Campbell Island



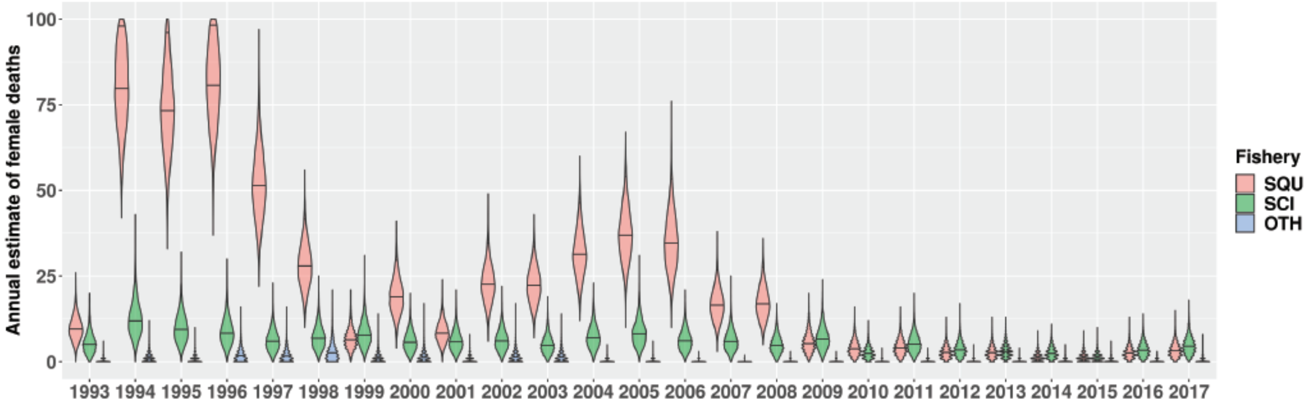


- Capture rates in relevant fisheries declined and stabilised after the full adoption of SLEDs (**see scheme above**). Observer coverage in squid fisheries has increased up to near 100%
- The Auckland Islands squid fishery is closed if the regulated mortality limit for NZ sea lions is reached

**Figures on left: fishing effort and observer coverage (above) and observed captures of NZ sea lion (below) in the Squid 6T trawl fishery 2003–2018**

# 4. SEA LION CAPTURE TRENDS IN TRAWL FISHERIES

- Sea lion captures are estimated for different trawl fisheries using a risk assessment model (see Chapter 3)



**Estimated NZ sea lion (females only) annual deaths 1993–2017 in trawl fisheries targeting squid (red); scampi (green); and all other trawls (blue) around the Auckland Islands**

- In the 1990s, relatively high captures are estimated to have occurred in trawl fisheries targeting squid. Estimated captures declined over time, as the fishing effort decreased
- From around 2007, sea lion deaths in the squid fishery decreased further, ranging between 1 and 5 deaths per year, after the universal adoption of standardised SLEDs
- Cryptic deaths, i.e., sea lions that exit via the SLED but nonetheless die as a consequence of the interaction, are estimated separately and included in the count of annual deaths (see figure above)

# 5. ONGOING RESEARCH

- Threat Management Plan (2017–2022) in place.
- Colony monitoring and pup counts are updated annually. SQU and SBW fisheries are highly observed. Spatial risk assessment can be updated annually using fishing overlap with sea lion distribution
- Work is in progress to monitor the new breeding sites along the Southland coast, and investigate indirect effects of fishing, diseases, and climate variability

# 4 NEW ZEALAND SEA LION (*PHOCARCTOS HOOKERI*)

| | |
|---|---|
| Status of chapter | This chapter has been substantially updated for AEBAR 2019–20, reflecting new science reviewed through the AEWG in 2019 and an updated management approach in the 2019 Squid 6T Operational Plan. |
| Scope of chapter | This chapter describes: the biology of New Zealand sea lions (NZSL; *Phocarctos hookeri*), the nature and extent of potential interactions with fisheries, means of estimating fisheries impacts and population-level risk, management of fisheries interactions, and priorities for future work. |
| Area | Auckland Islands, Campbell Island, and nearby sub-Antarctic waters over the continental shelf. Stewart Island and nearby coastal waters. Otago and the Catlins Coast and nearby coastal waters. |
| Focal localities | Areas with potential for significant fisheries interactions include the Auckland Islands Shelf, the Campbell Plateau, Stewart Island, and the southern and south-eastern coasts of the South Island. |
| Key issues | Improved understanding of the effects of fishing in the context of non-fishery threats and environmental variability; improved understanding of spatio-temporal distributions affecting interaction rates with fishing effort, with a focus on the Dundas Island and Figure of Eight Island breeding populations, and outside the summer season; improved understanding of the risk factors and population consequences of *Klebsiella pneumoniae*-infection and other causes of death for pups at the Auckland Islands and Campbell Island; improved understanding of the causes and population consequences of nutritional stress for the Auckland Islands and Campbell Island colonies; improved understanding of potential anthropogenic barriers to growth of South Island mainland and Stewart Island breeding populations; cryptic mortality in trawls employing Sea Lion Exclusion Devices (SLEDs). |
| Emerging issues | Improved means of estimating incidental captures and risk in poorly observed inshore fisheries potentially interacting with South Island and Stewart Island colonies. Improved understanding of the potential indirect effects of fishing on prey availability, in the context of climate variability. Management of public interactions with recovering South Island and Stewart Island populations. |
| Fisheries New Zealand research (current) | PRO2017-08C *Factors affecting New Zealand sea lion pup survival*; PMM2018-05B *Estimate spatial distributions for South Island NZSL to assess potential fisheries overlap and risk (including aquaculture)*. PMM2019-09: *Update Campbell Island NZSL PST (Population Sustainability Threshold) estimation;* ZBD2018-05: *Environmental variability, regime shifts, and ecosystem function in the sub-Antarctic*. |
| New Zealand government research (current) | DOC Marine Conservation Services Programme (CSP): INT2017-02 *Identification of marine mammals, turtles and protected fish captured in New Zealand fisheries*; INT2019-01 *Observing commercial fisheries*; INT2019-03 *Characterisation of marine mammal interactions*; POP2018-03 *New Zealand Sea Lion: Auckland Islands pup count;* MIT2014-01 *Protected species engagement project.* |
| Related chapters/issues | Chapter 5: New Zealand fur seals. |

## 4.1 CONTEXT

The management of fisheries impacts on New Zealand sea lions is legislated under the Marine Mammals Protection Act (MMPA) 1978 and the Fisheries Act (FA) 1996.

The Minister of Conservation gazetted the New Zealand sea lion as a threatened species in 1997. All marine mammal species are designated as protected species under s.2 (1) of the FA. In 2005, the Minister of Conservation approved the Conservation General Policy, which specifies in Policy 4.4 (f) that '*Protected marine species should be managed for their* *long-term viability and recovery throughout their natural range.'* The Department of Conservation's (DOC) Regional Conservation Management Strategies outline specific policies and objectives for protected marine species at a regional level. New Zealand's sub-Antarctic islands, including Auckland Islands and Campbell Island, were inscribed as a World Heritage area in 1998.

Fisheries New Zealand manages fishing-related mortality of New Zealand sea lions under s.15 (2) of the FA. Under that section, the Minister of Fisheries '*may take such measures as he or she considers are necessary to avoid, remedy, or mitigate the effect of fishing-related mortality on any protected species, and such measures may include setting a limit on fishing-related mortality.*'

The relevant National Fisheries Plan for the management of incidental captures of New Zealand sea lions is the National Fisheries Plan for Deepwater and Middle-depth Fisheries Part 1A (the National Deepwater Plan). Under the National Deepwater Plan, the objective most relevant for management of New Zealand sea lions is Environmental Outcome 8: *Manage deepwater and middle-depth fisheries to avoid, remedy, or mitigate the adverse effects of these fisheries on the long-term viability of endangered, threatened, and protected species*.

Specific objectives for the management of incidental captures of New Zealand sea lions will be outlined in the fishery-specific chapters of the National Deepwater Plan for the fisheries with which New Zealand sea lions are most likely to interact. These fisheries include sub-Antarctic trawl fisheries for arrow squid, southern blue whiting, and scampi.

The New Zealand sea lion population is monitored by pup counts at the main breeding colonies, the largest of which are on the Auckland Islands. The number of sea lion pups born at the Auckland Islands declined nearly 50% between 1998 and 2009 and appears to have stabilised thereafter. In 2014, following the third-lowest pup count on record, the Minister of Conservation and the Minister for Primary Industries requested that DOC and MPI work to develop a New Zealand sea lion/rāpoka Threat Management Plan (TMP). The process to develop the TMP involved a number of workstreams, including: a workshop to understand causes of pup mortality for sea lions at the Auckland Islands; two multi-day workshops, attended by a panel of independent experts, to inform a multi-threat risk assessment (Roberts 2015, Debski & Walker 2016); and inaugural meetings of the New Zealand sea lion/rāpoka Forum and Advisory Groups in early 2017. The TMP was finalised in 2017 (Department of Conservation & Ministry for Primary Industries 2017).

The TMP reflects the female New Zealand sea lion demographic population models and multi-threat risk assessment for the Auckland Islands described by Roberts & Doonan (2016), and recognises that no single identified threat in isolation was responsible for the population decline observed there since 2000. Population recovery would benefit from mitigation of multiple threats at the four main breeding sites (Department of Conservation & Ministry for Primary Industries 2017). The TMP commits to two objectives:

1) halt the decline of the New Zealand sea lion population within 5 years and
2) ensure the New Zealand sea lion population is stable or increasing within 20 years, with the ultimate goal of achieving 'Not Threatened' status.

The TMP outlines a work programme toward achievement of the plan's objectives, to be reviewed every five years. An overview of the TMP and identified workstreams, including research priorities, are reproduced in Figures 4.1 and 4.2.

## 4.2 BIOLOGY

### 4.2.1 TAXONOMY

The New Zealand sea lion (*Phocarctos hookeri*, Gray 1844) is one of only two species of otariid (eared seals, including fur seals and sea lions) native to New Zealand, the other being the New Zealand fur seal (*Arctocephalus forsteri*, Lesson 1828). The New Zealand sea lion is New Zealand's only endemic pinniped, in terms of the breeding distribution (noting that males haul out at Macquarie Island (an Australian sub-Antarctic island) but there is no breeding colony there).

### 4.2.2 HISTORICAL DISTRIBUTION

Before the arrival of humans in New Zealand, New Zealand sea lions ranged around the North and South islands of New Zealand and the Chatham Islands (Rawlence et al. 2016). Pre-European remains of New Zealand sea lions have been identified from at least 47 archaeological sites, ranging from Stewart Island to North Cape, with most occurring in the southern half of the South Island (Smith 1989, 2011, Childerhouse & Gales 1998, Gill 1998). Analysis of Holocene remains indicated that breeding sea lions once occurred

2017 – 2022

## New Zealand sea lion/rāpoka Threat Management Plan

**Vision:** Promote the recovery and ensure the long-term viability of New Zealand sea lions, with the ultimate goal of achieving 'Not Threatened' status.

**5 year objective:** Halt the decline of the New Zealand sea lion population within 5 years.

**20 year objective:** Ensure the New Zealand sea lion population is stable or increasing within 20 years.

### Site specific measures of success

**Stewart Island / Rakiura**

**Support population growth to achieve breeding colony status:**

a. Pup counts remain higher than 35 for 5 years in a row, qualifying this site as a new breeding colony

b. Pup production continues to increase to allow for colonial breeding

c. There are no cases of deliberate human-caused mortality

d. Public involvement in the conservation of sea lions increases.

**Auckland Islands**

**Pup production continues to increase from the 2014 count:**

a. Adult female survival rate and pup survival rate improve

b. Pup numbers are consistently above 1,575 (2014 pup count) and ideally over 1,965 (2017 pup count).

**South Island / Te Waipounamu**

**Support population growth:**

a. Pup counts along this stretch of coastline increase to above 16 per year, eventually reaching 35, on track to achieving breeding colony status

b. There are no cases of deliberate human-caused mortality

c. Public involvement in the conservation of sea lions increases.

**Campbell Island / Motu Ihupuku**

**Reduce pup mortality and support population growth:**

a. Pup counts are consistently at or above 696 (2015 pup count)

b. Pup mortality rates are consistently lower than 40% per annum, and

c. Frequency and consistency of monitoring of sea lions has increased.

### Success across the New Zealand sea lion range

1. Pup mortality from falling into natural holes is reduced
2. Disease research yields answers to inform recommendations to reduce pup mortality from *Klebsiella pneumoniae*
3. Estimation of SLED efficacy and cryptic mortality affecting adult female survival improves
4. The effects of climate change and fisheries on sea lion nutritional status are better understood
5. Sea lion breeding sites developing and colonies establishing at new locations
6. The New Zealand sea lion threat status improves from Nationally Critical to Not Threatened.

Figure 4.1: Threat management and population recovery objectives specific to four different New Zealand sea lion breeding populations, from the New Zealand sea lion Threat Management Plan (Department of Conservation & Ministry for Primary Industries 2017).

## 2017 – 2022
## New Zealand sea lion/rāpoka Threat Management Plan

**Vision:** Promote the recovery and ensure the long-term viability of New Zealand sea lions, with the ultimate goal of achieving 'Not Threatened' status.

**Partnership:** The principles of mātauranga Māori will be woven throughout all four workstreams to achieve the vision of the Threat Management Plan.

Workstreams

| | Engagement | Direct mitigation | Targeted research | Evaluation |
|---|---|---|---|---|
| | Coordinating and implementing an engagement campaign that facilitates a positive and accepted expansion of the range of sea lions. | Actions to reduce the impacts of key threats to sea lions are determined annually. | To address key threats to sea lions applied research will be determined annually. | To evaluate progress against the NZSL TMP objectives, monitoring of the sea lion breeding population is required. |
| 2017/2018 | • Establish the New Zealand sea lion/ rāpoka Forum and Advisory Group<br>• Establish a New Zealand sea lion community liaison officer to coordinate and implement a recovery strategy for South Island/ Te Waipounamu sea lions.<br>• Develop the broader engagment campaign | • 'Planks for Pups' programme at the Auckland Islands is maintained and expanded upon<br>• Develop a strategy to reduce pup mortality from natural holes at Campbell Island<br>• Establish a Technical Advisory Group to review the SQU6T Operational Plan | • Complete second season of disease research programme on the Auckland Islands<br>• Female sea lion nutritional stress and diet studies<br>• Research into the demographic parameters for at-risk marine mammals as identified by the marine mammal risk assessment (sea lions)<br>• Analysis of New Zealand sea lion tracking data to estimate overlap with fisheries<br>• Review of the potential impacts of aquaculture on sea lions, with relevance to Port Pegasus sea lion ecology | 2017/2018 Monitoring, including tagging, pup counts and tag re-sights, will occur at:<br>• Auckland Islands<br>• Campbell Island/Motu Ihupuku<br>• South Island/Te Waipounamu<br>• Stewart Island/Rakiura |
| 2018–2021 | Prioritisation of direct mitigation and targeted research undertaken annually based on the evaluation results and recommendations from the Forum and Advisory Groups. | | | |
| 2022 | Review against the NZSL TMP 20 year objectives | | | |

Figure 4.2: Workstreams identified in the New Zealand sea lion Threat Management Plan (Department of Conservation & Ministry for Primary Industries 2017).

around north-west Nelson, and that South Island and Chatham Island subpopulations were genetically distinct from contemporary New Zealand sea lions. These subpopulations became extinct shortly after the arrival of Polynesian settlers (Collins et al. 2014a, 2014b, Rawlence et al. 2016). Subsistence hunting on the South Island and subsequent commercial harvest from outlying islands of New Zealand sea lions for skins and oil resulted in population decline and contraction of the species range (Gales 1995, Childerhouse & Gales 1998, Nagaoka 2001, 2006). Despite the historic reduction in population size and range contraction as a result of subsistence hunting and commercial harvest, the New Zealand sea lion population does not display low genetic diversity at microsatellite loci and thus does not appear to have suffered effects of genetic drift and inbreeding depression (Robertson & Chilvers 2011).

### 4.2.3 CURRENT DISTRIBUTION

Currently, most New Zealand sea lions are found in the New Zealand sub-Antarctic, with individuals ranging to the New Zealand South Island and Macquarie Island. New Zealand sea lion breeding colonies[1] are highly localised, with most pups being born at the Auckland Islands and Campbell Island (Wilkinson et al. 2003, Chilvers 2008). At the Auckland Islands, there are three extant breeding colonies: Enderby Island (at Sandy Bay), Dundas Island, and Figure of Eight Island. On Campbell Island there is one breeding colony at Davis Point, another colony at Shoal Point, and an increasing number of non-colonial breeders (Wilkinson et al. 2003, Chilvers 2008, Maloney et al. 2009, Maloney et al. 2012; McNutt et al. 2020). Breeding on the Auckland Islands represents 68–79% of the pup production for the species, with the remaining 21–32% occurring on Campbell Island (based on concurrent pup counts in 2008, 2010, 2015, 2018, 2019, and 2020; see Figure 4.3). Numbers of breeding sea lions at the new Stewart Island colony and at haul-out sites on the South Island are comparatively low, but may be expected to increase steadily if these recolonisation events continue successfully.

Intermittent sea lion pup sightings have been reported at Port Pegasus, Stewart Island since the 1990s. In 2011, a pup survey and tagging programme was initiated, with 16 pups tagged. Breeding success at the Stewart Island location has increased steadily since that time, with 55 pups tagged in 2018 in the standard survey area, plus another 7 pups tagged outside the survey area (Boren 2018). In 2018, after 5 consecutive years with more than 35 pups being counted, the Stewart Island population was officially recognised as the third New Zealand sea lion breeding colony. The latest pup count for Stewart Island was 48 pups in 2020 (47 at Port Pegasus and one at Ulva Island; DOC unpublished data).

Successful sea lion breeding has also been observed on the Otago Coast, South Island, beginning with a single female that arrived in 1992 and gave birth in 1993 (McConkey et al. 2002). Pup production at this location increased slowly, to 7 pups in 2013, followed by a more rapid increase, to 21 pups in 2020 (see Figure 4.3).

On land, New Zealand sea lions are able to travel long distances and ascend hills. They are found in a variety of habitats including grass fields, exposed rock, and dense bush and forest; breeding colonies are usually on large sandy beaches (Gales 1995, Augé et al. 2012), though not everywhere, e.g., at Campbell Island. In early summer, colonial breeding sea lions are spatially constrained in the vicinity of colony locations. Following the end of the females' oestrus cycle in late January, adult and sub-adult males disperse throughout the species range, whereas the dispersal of females (both breeding and non-breeding) is more restricted both during and subsequent to the breeding season (Marlow 1975, Robertson et al. 2006, Chilvers & Wilkinson 2008).

### 4.2.4 FORAGING ECOLOGY

Foraging studies have been conducted on known populations of lactating female New Zealand sea lions, i.e., from Enderby Island, Dundas Island, and Figure of Eight Island (all in the Auckland Islands group) (Chilvers et al. 2005b, 2006, 2013, Chilvers & Wilkinson 2009); Stewart Island; and the Otago Peninsula (see Augé et al. 2011a, 2014, Chilvers et al. 2011). Leung et al. (2012, 2013b, 2014b) investigated foraging by juvenile New Zealand sea lions at Enderby Island in contrast with juvenile animals at Otago Peninsula (Leung et al. 2013a), and in mother-yearling pairs at Enderby Island (Leung et al. 2014a). A comprehensive analysis of spatial foraging patterns of Auckland Islands females used all the available satellite telemetry data to characterise spatial foraging patterns and estimate spatial overlap, annual deaths, and population risk

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[1] DOC (2009) defines colonies as 'haul-out sites where 35 pups or more are born each year for a period of 5 years or more.' Haul-out sites are defined as 'terrestrial sites where New Zealand sea lions occur but where pups are not born, or where fewer than 35 pups are born per year over 5 consecutive years.'

from all Auckland Islands commercial trawl fisheries (Large et al. 2019). This assessment estimated foraging distributions fthat primarily represent the summer foraging of breeding-age females from the Sandy Bay colony on Enderby Island. Further tracking is planned to collect data also from sea lions at Dundas Island and/or Figure of Eight Island, and to prioritise tracking data outside summer months. Analyses of satellite-tracked individuals from Campbell Island is in preparation (Lea et. al. in prep, Lea et al. in press.).

Previous analyses of sea lion foraging indicate that females from Enderby Island forage primarily over the Auckland Islands continental shelf and its northern edge, and that individuals show strong foraging site fidelity both within and across years. Satellite tagging data from lactating females at the Auckland Islands shows that the mean return distance travelled per foraging trip is 423 ± 43 km (n = 26), which is greater than that recorded for any other sea lion species (Chilvers et al. 2005b). While foraging, about half of the time was spent submerged, with a mean dive depth of 130 ± 5 m (max. 597 m) and mean dive duration of 4 ± 1 minutes (max. 14.5 minutes; Chilvers et al. 2006). Both juvenile female and male sea lions foraged to the north of the Auckland Islands, but the mean distance travelled per foraging trip was shorter in females (99 ± 12 km, n = 19) compared with males (184 ± 25 km, n = 12), and the mean maximum distance from the colony for males (93 ± 10 km) was about twice that for females (51 ± 5 km; Leung et al. 2012). A study of seven dependent yearling New Zealand sea lions (Leung et al. 2013b) found that dive depth was negatively related with animal mass (lighter sea lions dived to greater depths), but in juvenile (2–5 years old) New Zealand sea lions, diving ability (dive depth, dive duration, and bottom time per dive) improved with both mass and age, and five-year-old male New Zealand sea lions had similar dive capability to adult females (Leung et al. 2014b). New Zealand sea lions, like most pinnipeds, may use their whiskers to help them locate and capture prey at depths where light does not penetrate (Marshall 2008, Hankel et al. 2010). Leung et al. (2014a) found no evidence that yearling New Zealand sea lions were developing foraging skills through observational learning of maternal behaviours in a study of seven mother-yearling partnerships at Enderby Island.

A recent review of studies conducted on female New Zealand sea lions suggests a continuum of foraging behaviour between benthic foraging vs. mesopelagic foraging modes (Roberts et al. 2018; Lea et al. in press). An earlier study suggested that individual animals may tend to specialise in one or the other foraging mode (Chilvers & Wilkinson 2009). In that study benthic divers had fairly consistent dive profiles, reaching similar depths (120 m on average) on consecutive dives in relatively shallow water, presumably to feed on benthic prey. Mesopelagic divers, by contrast, exhibited more varied dive profiles, undertaking both deep (over 200 m) and shallow (less than 50 m) dives over deeper water. Benthic divers tended to forage further from their breeding colonies, making their way to the north-eastern limits of Auckland Islands Shelf, whereas mesopelagic divers tended to forage along the north-western edge of the shelf over depths of approximately 3000 m (Chilvers & Wilkinson 2009). Meynier et al. (2014), employed fatty acid (FA) analyses of blubber samples and found that FA profiles were different in primarily benthic diving vs. primarily mesopelagic diving lactating New Zealand sea lions, suggesting a different utilisation of prey resources such that, though prey species taken were similar for both dive modes, the proportion of particular prey differed between the two modes. In addition, Chilvers (2017) found that the composition of stable isotopes obtained from both blood serum and whiskers differed between benthic vs. mesopelagic foraging sea lions at the Auckland Islands. Further, Meynier et al. (2014) found that the body condition index (BCI: the residual between the measured and predicted body mass from the mass-length regression provided by Childerhouse et al. 2010a) was significantly greater in meso-pelagic divers than in benthic divers.

The differences in dive profiles have further implications for the estimated aerobic dive limits (ADL; Gales & Mattlin 1997, Chilvers et al. 2006), defined as the maximum amount of time that can be spent underwater without increasing blood lactate concentrations (a byproduct of anaerobic metabolism). If animals exceed their ADL and accumulate lactate, they must surface and go through a recovery period to aerobically metabolise the lactate before they can undertake subsequent dives. Chilvers et al. (2006) estimated that lactating female New Zealand sea lions at the Auckland Islands exceed their ADL on 69% of all dives, a much higher proportion than most other otariids (which exceed their ADL for only 4–10% of dives; Chilvers et al. 2006). Auckland Islands sea lions that exhibit benthic diving profiles are estimated to exceed their ADL on 82% of dives, compared with 51% for meso-pelagic divers (Chilvers 2008).

Chilvers et al. (2006) and Chilvers & Wilkinson (2009) suggested that the long, deep-diving behaviour, the propensity to exceed their estimated ADL, and differences in physical condition and age at first reproduction from animals at Otago together indicate that females from the Auckland Islands may be foraging at or near their physiological limits. However, Bowen (2012) suggested a lack of relationship between surface time and anaerobic diving would seem to indicate that ADL has been underestimated. Further, given a number of studies of diving behaviour were conducted during early lactation when the demands of offspring are less than they would be later in lactation, Bowen (2012) considered it unlikely that females are operating at or near a physiological limit.

Adult females at Otago are generally heavier for a given age, breed earlier, undertake shorter foraging trips, and have shallower dive profiles compared with females from the Auckland Islands (Table 4.1). These observed differences may reflect differences in habitat (including prey availability) between the Auckland Islands and the Otago Peninsula, or a founder effect, or a combination of these or other factors. Similarly, Leung et al. (2013a) compared foraging characteristics in juvenile (2–3 years old) female New Zealand sea lions at Enderby Island and Otago Peninsula. Overall, females at Otago were heavier (3 year old mean 96 kg) than females at Enderby (3 year old mean 72 kg), and exhibited shorter mean foraging trip distance (19 km at Otago, 103 km at Enderby), shallower mean dive depth (15 m at Otago, 69 m at Enderby) and shorter mean dive duration (1.8 min at Otago, 3.2 min at Enderby). Leung et al. (2013a) concluded that the Auckland Islands are a less optimal habitat compared with Otago. Satellite telemetry studies collected data during the 2019–20 summer field season to characterise the foraging distribution and dive behavior of breeding females in the Catlins coast mainland population (DOC unpublished data). Similar work may continue in future field seasons.

New evidence from satellite tracked individuals at Campbell Island (Lea et al. in press) and from analysis of sea lion prey including a dedicated ocean survey (Roberts et al. 2018) suggests that sea lions at the sub-Antarctic islands may suffer from periods of low prey availability and may be forced to forage at the limits of their physiological capabilities by low prey availability over the shelf. This would make these populations particularly susceptible to environmental variability affecting availability of preferred prey (Roberts et al. 2018).

The foraging of lactating females at Port Pegasus, Stewart Island was recently characterised by Chilvers (2018), describing their foraging characteristics as intermediate between Auckland Islands and Otago Peninsula females with respect to dive depth, dive duration, and body mass. Satellite telemetry data indicated that nearly all foraging was within 50 km of the tagging site at Port Pegasus (Roberts 2017a).

Table 4.1: Comparison of selected characteristics between adult female New Zealand sea lions from the Auckland Islands and those from the Otago Peninsula (Augé et al. 2011a, 2011b, Chilvers et al. 2006, Chilvers 2018, Roberts & Doonan 2016). Data are means ± s.e. (where available).

| Characteristic | Auckland Islands | Stewart Island | Otago Peninsula |
|---|---|---|---|
| Reproduction at age 4 | 19% of females<br>(95 % CI = 16 –23 %) | Unknown | > 85% of females |
| Average mass at 8–13 years of age | 112 kg | Unknown | 152 kg |
| Foraging distance from shore | 102.0 ± 7.7 km<br>(max = 175 km) | 45.0 ± 4.1 km<br>(max = 38 km) | 4.7 ± 1.6 km<br>(max = 25 km) |
| Time spent foraging at sea | 66.2 ± 4.2 hrs | 14.9 ± 1.4 hrs | 11.8 ± 1.5 hrs |
| Dive depth | 129.4 ± 5.3 m<br>(max = 597 m) | 59.6 ± 7.0<br>(max ≥ 250 m) | 20.2 ± 24.5 m<br>(max = 389 m) |
| Dives estimated to exceed ADL | 68.7 ± 4.4% | 35.1 ± 3.3% | 7.1 ± 8.1% |

New Zealand sea lions are generalist predators with a varied diet that includes marine mammal prey (New Zealand fur seal *Arctocephalus forsteri*), seabirds (yellow-eyed penguin *Megadyptes antipodes*, blue penguin *Eudyptula minor*, southern rockhopper penguin *Eudyptes chrysocome*, southern royal albatross *Diomedea epomophora*), elasmobranchs (rough skate *Raja nasuta*), teleost fish (e.g., opalfishes *Hemerocoetes* spp., hoki *Macruronus novaezelandiae*, red cod *Pseudophycis bachus*, jack mackerels *Trachurus* spp., barracouta *Thyrsites atun,* southen blue whiting *Micromesistius australis*); cephalopods (e.g., octopus *Enteroctopus zelandicus* and

*Macroctopus maorum*, squid *Nototodarus sloanii*); crustaceans (e.g., lobster krill *Munida gregaria*); and other invertebrates (e.g., salps) (Cawthorn et al. 1985, Moore & Moffat 1992, Bradshaw et al. 1998, Childerhouse et al. 2001, Lalas et al. 2007, Moore et al. 2008, Meynier et al. 2009, Augé et al. 2012, Lalas et al. 2014, Lalas & Webster 2014, Morrison et al. 2017). The three main methods used to assess New Zealand sea lion diets involve analyses of stomach contents, scats, and regurgitate, and the fatty acid composition of blubber (Meynier et al. 2008). Stomach contents of incidentally captured animals tend to be biased towards the target species of the fishery concerned (e.g., squid in the Auckland Islands squid fishery), whereas scats and regurgitates are biased towards less digestible prey (Meynier et al. 2008). Stomach, scat, and regurgitate approaches tend to reflect only recent prey (Meynier et al. 2008). By contrast, analysis of the fatty acid composition of blubber provides a longer-term perspective on diets ranging from weeks to months (although individual prey species are not identifiable). Fatty acid analysis suggests that the diet of female New Zealand sea lions at the Auckland Islands tends to include proportionally more arrow squid and hoki and proportionally fewer red cod than for male New Zealand sea lions, and that lactating and non-lactating females do not differ in their diet (Meynier et al. 2008, Meynier 2010). Within a sample of lactating female New Zealand sea lions, Meynier et al. (2014) used fatty acid analyses to show that the diet of benthic diving and meso-pelagic diving animals consisted of similar prey, though different mass contributions for each prey species.

Previous assessments have identified considerable spatial (comparing colonies) and temporal (inter-annual and seasonal) variation in the diet composition of New Zealand sea lions. For instance, jack mackerel and barracouta were identified as the main prey of the Otago Peninsula population (Augé et al. 2012), though were less prevalent in winter and spring when inshore species dominated diet composition (Lalas 1997) and were infrequent prey of the Auckland Islands population (Childerhouse et al. 2001, Stewart-Sinclair 2013). A long-term diet assessment of the Sandy Bay colony at the Auckland Islands (1994–95 to 2012–13) identified a decrease in the occurrence of large-sized prey (e.g., *Enteroctopus zealandicus*) and an increasing trend in small-sized prey (e.g., opalfishes, rattails, and *Octopus* spp.) (Childerhouse et al. 2001, Stewart-Sinclair 2013).

Teeth from individual sea lions at the Auckland Islands that were archived at Massey University and Te Papa Tongarewa were used to estimate trophic histories over an extended historical period. Graham et al. (2019) analysed 396 samples from the annual growth bands found in 22 sea lion teeth (19 females and 3 males) dating from 1935 to 2005. Nitrogen isotope (δ15N) data indicate an animal's trophic ecology and changes in their foraging strategies. It was found that the male sea lions consistently forage at a higher trophic level than the females. The δ15N values of the 19 females reveal aspects of their foraging ecology and physiology. At a broad scale, there is considerable variation between individuals, suggesting variable foraging strategies. A maternal or lactation signal was observed in almost all teeth samples. This signal occurs because as the pup consumes the mother's milk its isotope value will be one trophic level higher than its mother. In general, the lactation signal declined for most of the individuals in the first year, and by year two it was only present in two individuals born in 1943 and 1994. An increase in trophic level occurs after age five, which coincides with the age at first breeding, but again there is inter-individual variation. The δ13C dataset for female New Zealand sea lions shows an overall decreasing temporal trend, with notable decreases pre-1960 and post-1990. Changes in primary productivity affect the δ13C values at the base of the food web and this signal has been shown to propagate up the food web. Overall, in periods of higher productivity the δ13C values increase (Laws et al. 1995, Schell et al. 1998, Graham et al. 2010). This suggests that during the 1940–60s and late 1990–early 2000s there was either a) a decrease in productivity around the Auckland Islands where the female sea lions forage (i.e., shift in ocean conditions) and/or b) the females shifted their main foraging strategy (e.g., benthic vs. mesopelagic related to available prey). A higher sample size would be required from the earlier time period to resolve the timing of these isotopic signals because they may relate to ecosystem changes potentially affecting fish stocks or other species.

### 4.2.5 REPRODUCTIVE BIOLOGY

New Zealand sea lions exhibit marked sexual dimorphism; adult males are darker in colour and much larger than adult females (Walker & Ling 1981, Cawthorn et al. 1985). Cawthorn et al. (1985) and Dickie (1999) estimated the maximum age of males and females to be 21 and 23 years, respectively; Childerhouse et al. (2010b) reported a maximum estimated age for females of 28 years. Females can become sexually mature as early as age two and may give birth the following year. However, at the Auckland Islands most females do not breed until they are six years old (Childerhouse et al. 2010b; Roberts & Doonan 2016); at

Otago Peninsula most females breed by age four (Roberts & Doonan 2016). Males generally reach sexual maturity at age four, but because of their polygynous colonial breeding strategy (i.e., males actively defend territories and mate with multiple females within a harem) they are only able to successfully breed at 7–9 years old, once they have attained sufficient physical size to compete successfully with other males (Marlow 1975, Cawthorn et al. 1985). At the Auckland Islands, the reproductive rate in females increases rapidly between the ages of 3 and 7, reaching a plateau until the age of approximately 15 and declining rapidly thereafter, with the maximum recorded age at reproduction being 26 years (Breen et al. 2016, Childerhouse et al. 2010a, Chilvers et al. 2010). Chilvers et al. (2010) estimated from tagged sea lions that the median lifetime reproductive output of a female New Zealand sea lion at the Auckland Islands was 4.4 pups, and 27% of all females that survive to age 3 never breed. Analysis of tag-resighting data from female New Zealand sea lions on Enderby Island indicates the average probability of breeding is approximately 0.30–0.35 for prime-age females that did not breed in the previous year (ranges reflect variation relating to the definition of breeders) and 0.65–0.68 for prime-age females that did breed in the previous year (MacKenzie 2011).

New Zealand sea lions are strongly philopatric (i.e., they return to breed at the same location where they were born, although more so for females than males). Breeding is highly synchronised and starts in late November when adult males establish territories (Robertson et al. 2006, Chilvers & Wilkinson 2008). Pregnant and non-pregnant females appear at the breeding colonies in December and early January, with pregnant females giving birth to a single pup in late December before entering oestrus 7–10 days later and mating again (Marlow 1975). Twin births and the fostering of pups in New Zealand sea lions are rare (Childerhouse & Gales 2001). Shortly after the breeding season ends in mid-January, the harems break up with the males dispersing offshore and females often moving away from the rookeries with their pups (Marlow 1975, Cawthorn et al. 1985).

Pup birth weight is 8–12 kg and is highly variable between years; parental care is restricted to females (Walker & Ling 1981, Cawthorn et al. 1985, Chilvers et al. 2006). Females remain ashore for about ten days after giving birth before alternating between foraging trips lasting approximately two days at sea and returning for about one day to suckle their pups (Gales & Mattlin 1997, Chilvers et al. 2005b). New Zealand sea lion pup growth rates at the Auckland Islands are lower than those reported for other sea lion species and may be linked to a relatively low concentration of lipids in the females' milk during early lactation (Chilvers 2008, Riet-Sapriza et al. 2012). Riet-Sapriza et al. (2012) also found that there was a temporal (year and month) effect on milk quality, reflecting individual sea lion characteristics and environmental factors, and that maternal body condition was positively correlated with milk lipid concentration, energy content, and milk protein concentration: lactating females in good condition produced more energy-rich milk than did relatively lean females. Pups are weaned after about 10–12 months (Marlow 1975, Gales & Mattlin 1997).

### 4.2.6 POPULATION BIOLOGY

For New Zealand sea lions, the overall size of the population is indexed using estimates of the number of pups that are born each year (Chilvers et al. 2007). Moderately reliable pup counts have been made at Auckland Islands colonies since the 1960s (e.g., Falla et al. 1979, and see a review by Childerhouse & Gales 1998, and summary of estimates in table 1 of Breen et al. 2016), though these were intermittent, and reliable counts were not made across all known Auckland Islands colonies in the same year prior to 1995. Since 1995, the Department of Conservation (DOC) has conducted mark-recapture and pup census counts at each of the main breeding colonies at the Auckland Islands, using a consistent methodology, to estimate annual pup production (i.e., the total number of pups born each year, including dead and live animals; Robertson & Chilvers 2011). Pup censuses have been less frequent for other colonies, including the large population at Campbell Island (Maloney et al. 2012).

For the Auckland Islands population, the data show a decline in pup production from a peak of 3021 in 1997–98 to a low of 1501 ± 16 pups in 2008–09 (Chilvers & Wilkinson 2011, Robertson & Chilvers 2011; see Figure 4.3 and Table 4.2), with the largest single-year decline (31%) occurring between the 2008 and 2009 counts.

Since 2009, estimated pup production at the Auckland Islands appears to have stabilised, fluctuating without trend between roughly 1600 and 1800 pups in most years. The most recent estimate of pup production for the Auckland Islands population was 1740 pups in 2020, of which 289 were at Sandy Bay and 1399 were at Dundas Island (Table 4.2 and Figure 4.3).

The total New Zealand sea lion population size (including pups) at the Auckland Islands has been estimated using

Bayesian population models (Breen et al. 2003, 2016, Breen & Kim 2006a, 2006b, Roberts & Doonan 2016). Although other abundance estimates are available (e.g., Gales & Fletcher 1999), for the Auckland Islands population, estimates derived from the integrated models are preferred because they take into account a variety of age-specific factors (breeding, survival, maturity, incidental fisheries captures), as well as data on the resighting of tagged animals and pup production estimates (Table 4.3).

When using demographic models to predict future population trends, the future trajectory of the Auckland Islands population is highly dependent on the time period of demographic rates used to generate forward projections. For instance, negative population growth ($\lambda = 0.959$; 95% credible interval = 0.955–0.963) was estimated when using the demographic rates for the period of declining pup production (between 1999 and 2009). But increasing ($\lambda = 1.087$; 95% credible interval = 1.069–1.105) or approximately stable ($\lambda = 0.989$; 95% credible interval = 0.985–0.993) trajectories were produced when using demographic rates for the prior period of growth (until 1999) or relatively stability (since 2009) was used (Roberts 2019).

At locations outside the Auckland Islands, breeding sea lions have only established more recently, so their expected population trajectories may be generally characterised by initial population growth followed by eventual stabilisation as populations approach local habitat limits, or by alternating periods of population growth and decline reflecting variable environmental conditions.

At Campbell Island, recorded pup production has grown from very low levels in the early 1990s up to 734 pups in 2018 (Boren 2018) and was 595 pups in 2020 (although note that comparability may have been affected by exceptionally high pup mortality rate in the latest year; see below) (McNutt et al. 2020). Estimates of pup production at Campbell Island increased sharply in the period from 1990 to 2010 (i.e., including during the period of steepest decline at the Auckland Islands) but there has been some variation in the timing and methodology of these surveys, and one of the breeding colonies has moved over time. The later surveys in 2003, 2008, 2010, and 2015 were considered to be of sufficient quality to inform a simple population estimate (Roberts & Doonan 2016) and a comparable methodology was used to estimate pup production in 2018, 2019, and 2020. Early pup mortality (i.e., in the first few months of life) at Campbell Island has been relatively high in all recent census years, including: 1998 (31%), 2003 (36%), 2008 (40%), 2010 (55%), 2015 (58%), 2018 (23%), 2019 (54%), and 2020 (81%, the highest recorded at any New Zealand sea lion breeding site) (see Childerhouse et al. 2005, 2015a, Boren 2018, Foo & Weir 2019, Maloney et al. 2009, 2012, McNally et al. 2001, McNutt et al. 2020). Multiple consecutive years of high pup mortality rates can be expected to have detectable impacts on adult population size and future pup production as the affected cohorts reach maturity and recruit into the breeding population.

For the Otago coast, annual pup production has increased from 0 in the 1995 breeding season to 21 in the 2020 season (Figure 4.3). Sea lions at Otago are of special interest because they highlight the potential for establishing new breeding colonies; the Otago coast breeding population originated with a single pregnant female (McConkey et al. 2002). The TMP identifies that the viability of new colony locations on the New Zealand South Island is of particular importance for the restoration of New Zealand sea lions to non-threatened status.

Sea lions have established at Stewart Island, where pup census estimates have been made since 2011, about 3–4 months after the probable pupping period. Stewart Island pup counts have increased from 16 pups in 2011 to 48 pups in 2020 (Chilvers 2014, DOC unpublished data, Roberts & Doonan 2016; Figure 4.3). From 2018, the Stewart Island population was formally recognised as a new breeding colony, after 5 consecutive years in which annual pup production was estimated to have exceeded 35 births (Department of Conservation & Ministry for Primary Industries 2017).

### 4.2.7 THREATS TO SEA LIONS

Known anthropogenic sources of direct mortality to New Zealand sea lions include, historically: subsistence hunting and commercial harvest (Gales 1995, Childerhouse & Gales 1998); and pup entrapment in rabbit burrows prior to rabbit eradication from Enderby Island in 1993 (Gales & Fletcher 1999). On Stewart Island and the South Island sea lions encounter human disturbance (including attacks by dogs), vehicle strikes, and deliberate shooting on South Island New Zealand (Gales 1995). Incidental captures in fisheries may affect both sub-Antarctic and mainland populations (see section 4.4). Scientific research may also pose a threat, e.g., there is a risk of accidental death arising from the use of anesthesia (Lynch et al. 1999), and disturbance and handling of animals may create other risks to animal health. Other anthropogenic effects may indirectly affect New Zealand sea lion populations, but for which the actual level of impact is presently unclear, include potential trophic competition between New Zealand sea lions and fisheries (Robertson & Chilvers 2011, Bowen 2012, Roberts et al. 2018; see below); effects of organic and inorganic pollutants, including polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDT), and heavy metals such as mercury and cadmium (Baker 1999, Robertson & Chilvers 2011); and impacts of casual or organized eco-tourism.

Very high rates of pup mortality observed at Campbell Island are mainly due to pups drowning in wallows (Lea et al. 2018), or dying of exposure or starvation arising from adverse weather conditions (McNutt et al. 2020). The magnitude of these impacts may reflect that a substantial proportion of Campbell Island sea lions may be breeding in locations with sub-optimal conditions (i.e., muddy beach substrate and high exposure to extreme weather events). Similarly, high rates of pup mortality from holes and storms are not typically observed elsewhere. Other sources of natural mortality that may occur in all locations include predation by white pointer sharks (Cawthorn et al. 1985, Robertson & Chilvers 2011), starvation of pups if they become separated from their mothers (Walker & Ling 1981, Castinel et al. 2007), and male aggression towards females and pups (Chilvers et al. 2005a; Wilkinson et al. 2000).

#### 4.2.7.1 DISEASE

Epizootic diseases can be a significant threat to New Zealand sea lion populations; for example *Campylobacter*, which is thought to have killed 1600 pups (53% of pup production) and at least 74 adult females on the Auckland Islands in 1997–98 (Wilkinson et al. 2003, Robertson & Chilvers 2011).

More recently, *Klebsiella pneumoniae* killed 33% and 21% of new pups at the Auckland Islands in 2001–02 and 2002–03, respectively (Wilkinson et al. 2006) and 55% of pups between 2009 and 2014 (Roe et al. 2014). A hypermucoviscous (highly-sticky) strain of *K. pneumoniae* was isolated from a number of pups that died in field seasons 2005–06 to 2009–10 (Roe 2011). In this period, disease-related mortalities occurred late in the field season relative to the period 1998–99 to 2004–05 and were still occurring up to the end of sampling (Castinel et al. 2007, Roe 2011). *K. pneumoniae* was found to have caused on average 60% of pup deaths annually at Enderby Island between 2013 and 2018 (Table 4.4, with likely more continuing mortality following pup dispersal and the cessation of the summer monitoring season (Michael et al. 2019)). By comparison, less common causes of pup death over this time included starvation (14.8%), trauma or asphyxiation (9.9%), and other infections (7%) (Michael et al. 2019).

The 1998 epizootic event may also have affected the fecundity of the surviving pups, reducing their breeding rate relative to other cohorts (Gilbert & Chilvers 2008), though the pupping rate estimate for this cohort is likely to have been negatively biased by particularly high tag shedding rates for individuals tagged in that year (Roberts et al. 2014a).

Table 4.2: Pup census estimates for all known breeding populations of New Zealand sea lions since 1994–95. Years with no census estimates were left blank (i.e., blanks do not necessarily indicate that no pups were born at that location in that year). See table 1 of Breen et al. (2016) for a summary of counts from years prior to 1990 and the review by Childerhouse & Gales 1998. (Continued on next page)

| Pupping season | Annual pup census estimate | | | | | |
|---|---|---|---|---|---|---|
| | Auckland Islands | | | Campbell Island | Otago coast | Stewart Island |
| | Dundas Island | Sandy Bay | All | | | |
| 1990 | | 434 | | | | |
| 1991 | | 429 | | | | |
| 1992 | 1 934 | 489 | | | | |
| 1993 | 1 870 | 432 | | | | |
| 1994 | | | | | | |
| 1995 | 1 837 | 467 | 2 518 | | 0 | |
| 1996 | 2 017 | 455 | 2 685 | | 1 | |
| 1997 | 2 260 | 509 | 2 975 | | 0 | |
| 1998 | 2 373 | 477 | 3 021 | | 2 | |
| 1999 | 2 186 | 513 | 2 867 | | 1 | |
| 2000 | 2 163 | 506 | 2 856 | | 1 | |
| 2001 | 2 148 | 562 | 2 859 | | 3 | |
| 2002 | 1 756 | 403 | 2 282 | | 3 | |
| 2003 | 1 891 | 488 | 2 516 | 385 | 3 | |

| Pupping season | Annual pup census estimate | | | | | |
|---|---|---|---|---|---|---|
| | Auckland Islands | | | Campbell Island | Otago coast | Stewart Island |
| | Dundas Island | Sandy Bay | All | | | |
| 2004 | 1 869 | 507 | 2 515 | | 3 | |
| 2005 | 1 587 | 441 | 2 148 | | 4 | |
| 2006 | 1 581 | 422 | 2 089 | | 7 | |
| 2007 | 1 693 | 437 | 2 224 | | 4 | |
| 2008 | 1 635 | 448 | 2 175 | 583 | 6 | |
| 2009 | 1 132 | 301 | 1 501 | | 5 | |
| 2010 | 1 369 | 385 | 1 814 | 681 | 6 | |
| 2011 | 1 089 | 378 | 1 550 | | 6 | 16 |
| 2012 | 1 248 | 361 | 1 684 | | 6 | 25 |
| 2013 | 1 491 | 374 | 1 940 | | 6 | 26 |
| 2014 | 1 213 | 290 | 1 575 | | 4 | 32 |
| 2015 | 1 230 | 286 | 1 576 | 696 | 8 | 36 |
| 2016 | 1 347 | 321 | 1 727 | | 15 | 31 |
| 2017 | 1 549 | 349 | 1 965 | | 16 | 41 |
| 2018 | 1 397 | 332 | 1 792 | 734 | 18 | 55 |
| 2019 | 1 295 | 319 | 1 679 | 705 | 17 | 52 |
| 2020 | 1 398 | 289 | 1 740 | 595 | 21 | 48 |

Table 4.3: Pup production and population estimates of New Zealand sea lions from the Auckland Islands. Pup production data are direct counts or mark-recapture estimates from Chilvers et al. (2007), Robertson and Chilvers (2011), Chilvers (2012a), and Childerhouse et al. (2014, 2015b, 2016), noting that counts of dead pups began later in 2013 and 2014 and this is likely to have led to a negative bias in estimates for these years. Standard errors apply only to the portion of pup production estimated using mark-recapture methods. Mature female and total female population estimates are from the base case model by Roberts (2019). Year refers to the second calendar year of a breeding season (e.g., 2010 refers to the 2009–10 season). (Continued on next page)

| Year | Pup production estimate | | Mature female population size | | Total female population size | |
|---|---|---|---|---|---|---|
| | Mean | Standard error (for mark recapture estimates)* | Median | 95% confidence interval | Median | 90% confidence interval |
| 1995 | 2 518 | 21 | 3 151 | 2 834–3 505 | 6 920 | 6 373–7 552 |
| 1996 | 2 685 | 22 | 3 369 | 3 067–3 703 | 7 027 | 6 559–7 560 |
| 1997 | 2 975 | 26 | 3 602 | 3 317–3 913 | 7 183 | 6 793–7 622 |
| 1998 | 3 021 | 94 | 3 819 | 3 559–4 106 | 7 363 | 7 034–7 723 |
| 1999 | 2 867 | 33 | 3 976 | 3 746–4 232 | 7 544 | 7 247–7 848 |
| 2000 | 2 856 | 43 | 4 098 | 3 889–4 328 | 7 591 | 7 269–7 929 |
| 2001 | 2 859 | 24 | 3 817 | 3 640–4 013 | 7 218 | 6 925–7 515 |
| 2002 | 2 282 | 34 | 3 582 | 3 426–3 755 | 6 863 | 6 598–7 124 |
| 2003 | 2 518 | 38 | 3 391 | 3 253–3 545 | 6 536 | 6 294–6 767 |
| 2004 | 2 515 | 40 | 3 239 | 3 114–3 381 | 6 233 | 6 009–6 445 |
| 2005 | 2 148 | 34 | 3 096 | 2 978–3 231 | 5 949 | 5 740–6 146 |
| 2006 | 2 089 | 30 | 2 952 | 2 839–3 079 | 5 662 | 5 481–5 833 |
| 2007 | 2 224 | 38 | 2 813 | 2 704–2 936 | 5 390 | 5 224–5 555 |
| 2008 | 2 175 | 44 | 2 688 | 2 581–2 808 | 5 129 | 4 966–5 295 |
| 2009 | 1 501 | 16 | 2 578 | 2 473–2 692 | 4 931 | 4 769–5 101 |
| 2010 | 1 814 | 36 | 2 484 | 2 379–2 596 | 4 786 | 4 617–4 961 |
| 2011 | 1 550 | 41 | 2 466 | 2 373–2 571 | 4 733 | 4 575–4 898 |
| 2012 | 1 684 | 22 | 2 444 | 2 354–2 545 | 4 681 | 4 530–4 834 |
| 2013** | 1 940 | 50 | 2 416 | 2 328–2 517 | 4 626 | 4 479–4 774 |
| 2014** | 1 575 | 19 | 2 384 | 2 292–2 486 | 4 569 | 4 424–4 717 |
| 2015 | 1 576 | | 2 355 | 2 262–2 457 | 4 512 | 4 363–4 661 |
| 2016 | 1 727 | | 2 327 | 2 232–2 428 | 4 456 | 4 304–4 610 |
| 2017 | 1 965 | | 2 299 | 2 202–2 402 | 4 401 | 4 242–4 561 |
| 2018 | 1 792 | | 2 271 | 2 172–2 377 | 4 346 | 4 181–4 517 |

| Year | Pup production estimate | | Mature female population size | | Total female population size | |
|---|---|---|---|---|---|---|
| | Mean | Standard error (for mark recapture estimates)* | Median | 95% confidence interval | Median | 90% confidence interval |
| 2019 | 1 679 | | 2 244 | 2 141–2 355 | 4 293 | 4 120–4 473 |
| 2020 | 1 740 | | | | | |

* Calculated as the sum of standard errors associated with estimates for Sandy Bay and Dundas Island (estimates for other rookeries from direct count rather than mark-recapture).

** Field season began later in these years and pups that died early in the pupping period were unlikely to have been included in pup production estimates.

*** Roberts & Doonan (2016) estimated 11 755 for the entire species.

Table 4.4: Annual proportions of necropsied New Zealand sea lion (*Phocarctos hookeri*) pups at Sandy Bay, Enderby Island at the Auckland Islands that were attributed to *Klebsiella pneumoniae* infection, for field seasons between 2013–14 and 2017–18 (Michael et al. 2019).

| Field season | Total necropsies | Number (and percentage) attributed to *K. pneumoniae* infection |
|---|---|---|
| 2013–14 | 69 | 48 (70%) |
| 2014–15 | 58 | 41 (71%) |
| 2015–16 | 33 | 13 (39%) |
| 2016–17 | 75 | 40 (53%) |
| 2017–18 | 49 | 29 (59%) |
| Total | 284 | 171 (60%) |

#### 4.2.7.2 INDIRECT/ TROPHIC EFFECTS OF FISHING

It is possible that indirect fisheries effects may have population-level consequences for New Zealand sea lions. Such indirect effects may include competition for food resources between various fisheries and New Zealand sea lions (Robertson & Chilvers 2011; Roberts et al. 2018). To determine whether resource competition is present and is having a population-level effect on New Zealand sea lions, research has sought to identify if there are resources in common for New Zealand sea lions and the various fisheries within their preferred foraging range, and to what extent those resources are limiting. Diet studies have revealed some overlap in the species consumed by New Zealand sea lions and those caught in fisheries within the range of New Zealand sea lions, particularly hoki and arrow squid (Cawthorn et al. 1985, Childerhouse et al. 2001, Meynier et al. 2009). Meynier et al. (2014) analysed energy and amino acid content of prey and determined that the selected prey species contained all essential amino acids and were of low to medium energy levels. This study concluded that given low energy densities of prey, sea lions may be able to sustain energy requirements, but not necessarily store energy reserves and, thus, sea lions may be sensitive to factors that negatively affect trophic resources. Meynier (2010) also developed a bio-energetic model and used it to estimate that roughly 17 871 t of prey are consumed by New Zealand sea lions at per year. This is about 30% of the annual harvest of arrow squid, and about 15% of the hoki harvested annually by the fisheries in the sub-Antarctic between 2000 and 2006; note however that later research suggests that squid and hoki do not constitute the major portion of sea lion diet (Roberts et al. 2018)

Comparison of the temporal and spatial distributions of sea lion prey, sea lion foraging, and of historical fishing extractions may help to identify the mechanisms whereby resource competition might occur (Bowen 2012), but the potential trophic effects of fishing on sea lions are likely to be complicated due to complex food web interactions. Multi-species models may help to assess the extent to which resource competition can impact on sea lion populations. Roberts et al. (2018) investigated the abundance and distribution of sea lion prey species, including via a dedicated trawl survey, and suggested that the Auckland Islands sea lion subpopulation has endured a protracted period of nutritional stress, such that during unfavourable periods this population may have been limited by the availability of key prey. However, conclusions regarding the extent to which this may reflect indirect fisheries effects are inconclusive pending a more thorough understanding of sea lion diet and foraging behaviour under different environmental conditions, and the relative impacts of other threats including episodic climate-induced mortality events and/or disease.

#### 4.2.7.3 CLIMATIC AND/OR FISHERIES-RELATED DRIVERS OF NUTRITIONAL LIMITATION

Temporally coincident changes in annual abundance, spatial distribution, and/or reproductive success have been observed in different ecosystem components at sub-Antarctic latitudes — including New Zealand sea lions, Antipodean albatrosses, and demersal and pelagic fish communities. These observations suggest that climatic variability at decadal scales (sometimes labeled 'regime shifts') may affect ecosystem productivity in these systems, in turn affecting critical demographic rates for sub-Antarctic islands sea lions and resulting decadal-scale population dynamics (see below). For instance, a retrospective analysis of trawl survey data from the Campbell Plateau found evidence for a decade-long period of very low hoki abundance at depths foraged by New Zealand sea lions (Roberts et al. 2018). Long-term shifts in the catch rates of arrow squid around the Auckland Islands and the Stewart-Snares shelves were estimated from commercial trawl data and appear to be correlated with changes in primary production through time (Hurst et al. 2012).

Fisheries New Zealand is progressing new research (project ZBD2018-05) to investigate the evidence for regime shifts in the marine ecosystem of the sub-Antarctic and adjacent areas, likely climatic drivers, and potential ecological consequences as reflected in the productivity of fish and megafauna species. This project aims to identify climate and productivity indices for monitoring changes to the ecosystem that affect New Zealand sea lions and other focal ecosystem components.

### 4.2.8 RELATING DEMOGRAPHIC RATES TO DRIVERS OF POPULATION CHANGE

Over several years, various demographic assessments have been conducted to identify the proximate demographic causes of observed population and pup production trends at the Auckland Islands (see MacKenzie 2011, Roberts et al. 2014a, Roberts and Doonan 2016, Roberts 2017b, Roberts 2019). Roberts et al. (2014a) concluded that that the substantial decline in pup production between 1999 and 2009 was a consequence of low pupping rates during this period (including occasional years with very low rates), a declining trend in cohort survival to age two (pup survival) and to age five (juvenile survival) since the early 1990s, and relatively low adult survival (age 6–14) from 1999–2000 to 2010–11 (Figure 4.4.) In particular, very low pup survival rates at the Sandy Bay colony in 2005–2007, if they are indicative of similar processes occurring also at other breeding locations (i.e., Dundas Island and Figure of Eight Island), are likely to have compromised breeder numbers and pup production in later years (Roberts & Doonan 2016).

The subsequent change in the observed pup production trajectory (from declining in 1999–2009 to stable in 2009–2019) appears to have been driven by increased juvenile survival (ages 2–5) and increased adult survival (ages 6–14) and by a slightly increased pupping rate, rather than by a significant increase in pup survival, which remains lower than was observed before the period of population decline (Roberts 2019). This work suggests that further improvement in the observed trend may not be possible without an increase in pup survival rates. The extent to which pup survival is affected by the disease *K. pneumoniae* (see below) on an ongoing basis remains a priority for future research. New modelling work was completed in 2020 (Edwards & Roberts in review; Roberts et al. in review) to evaluate the extent to which pup morphometric data (including mass and condition), collected annually by the DOC field team at the Auckland Islands, can be used as a covariate to improve demographic model predictions and better understand potential drivers of observed demographic rates affecting population trends (e.g., first year survival of pups and annual pupping rate). A correlative assessment was conducted to identify the causes of varying demographic rates at Sandy Bay, for which hypothetical models developed with expert consultation were used as a framework for testing relationships between demographic rate estimates, biological observations (e.g., diet composition, maternal body condition, or pup mass) and candidate drivers of population change (e.g., changes in prey availability, disease-related pup mortality, or direct fishery-related mortalities) (Roberts & Doonan 2014).

Climate indices including Interdecadal Pacific Oscillation (IPO) and sea surface height (SSH) were well correlated with the occurrence of an array of key prey species in scats (Childerhouse et al. 2001, Stewart-Sinclair 2013). A weak, though significant, positive correlation was identified between maternal body condition and pup mass in seasons from 1990–91 to 2004–05. In this time period, pup mass at three weeks appeared to have been a good predictor of cohort-specific survival to age two, though there was no relationship with cohorts born 2004–05 to 2009–10, for which survival estimates were consistently low despite high pup mass (Figure 4.5). A correlation between cohort survival to age two and the rate of pup mortalities attributed to *K. pneumonia* infection late in the field season (Castinel et al. 2007, Roe 2011) was consistent with disease-related mortality affecting a decline in pup/yearling survival after 2004–05. Survival at ages 2–5 y (juveniles) or ages 6–14 y (adults) were not correlated with the estimated level of fishery interactions in the Auckland Islands arrow squid (SQU 6T) trawl fishery (Thompson et al. 2011). However, from 1998–99 to 2003–04 survival at ages 6–14 y was negatively correlated with the survival of pups born in the previous year, suggesting that the high energetic costs of lactation may compromise maternal survival (Roberts & Doonan 2014).

In most cases observations were available only for short time periods and longer series would be required to identify a causative relationship. However, broad changes in diet composition (e.g., an increased prevalence of small-sized prey species), reduced maternal body condition, and depressed pupping rates, are all consistent with a sustained period of nutritional stress negatively affecting the productivity of New Zealand sea lions at the Auckland Islands.

In addition, disease-related mortality of pups since 2005–06 (Roe 2011) has caused a decline in pup/yearling survival, which may further compromise breeder numbers at the Auckland Islands in the immediate future. It has been suggested that nutritional stress can be expected to predispose the population to higher rates of disease mortality, such that pups in poorer condition may be expected to have higher rates of death from disease. However, preliminary blubber depth data up to the 2014 field season indicated that pups dying from *Klebsiella* infection were in comparatively good condition relative to pups dying from other causes (preliminary data from W. Roe, unpublished, see Figure 4.6). As such, if poor body condition once predisposed pups to death from *Klebsiella* infection, then this no longer appears to be so. The relative contributions of *Klebsiella* infection, nutritional stress, or other factors potentially affecting first year mortality of pups are unclear, these are being investigated in ongoing research (e.g., Edwards & Roberts in review, Roberts et al. in review).

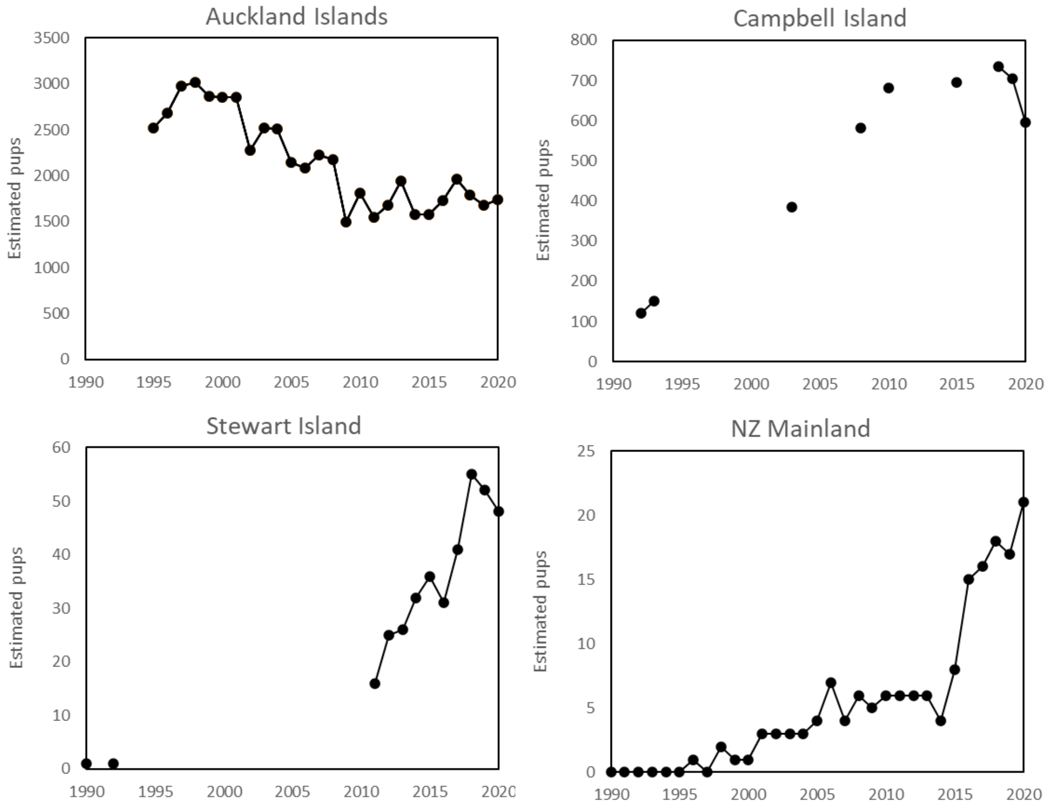

Figure 4.3: Annual sea lion pup count estimates from breeding sites (DOC unpublished data, McNutt et al. 2020, Melidonis & Childerhouse 2020). Note that the y-axis scale is different in each figure. Where count methodology was not consistent between adjacent years, annual point estimates are not joined by solid lines in the figures (e.g., some years in the Stewart Island and Campbell Island figures). Note that because the location of the Campbell Island breeding colony appears to be changing, it is possible that the 2020 pup production estimate is biased low (McNutt et al. 2020).

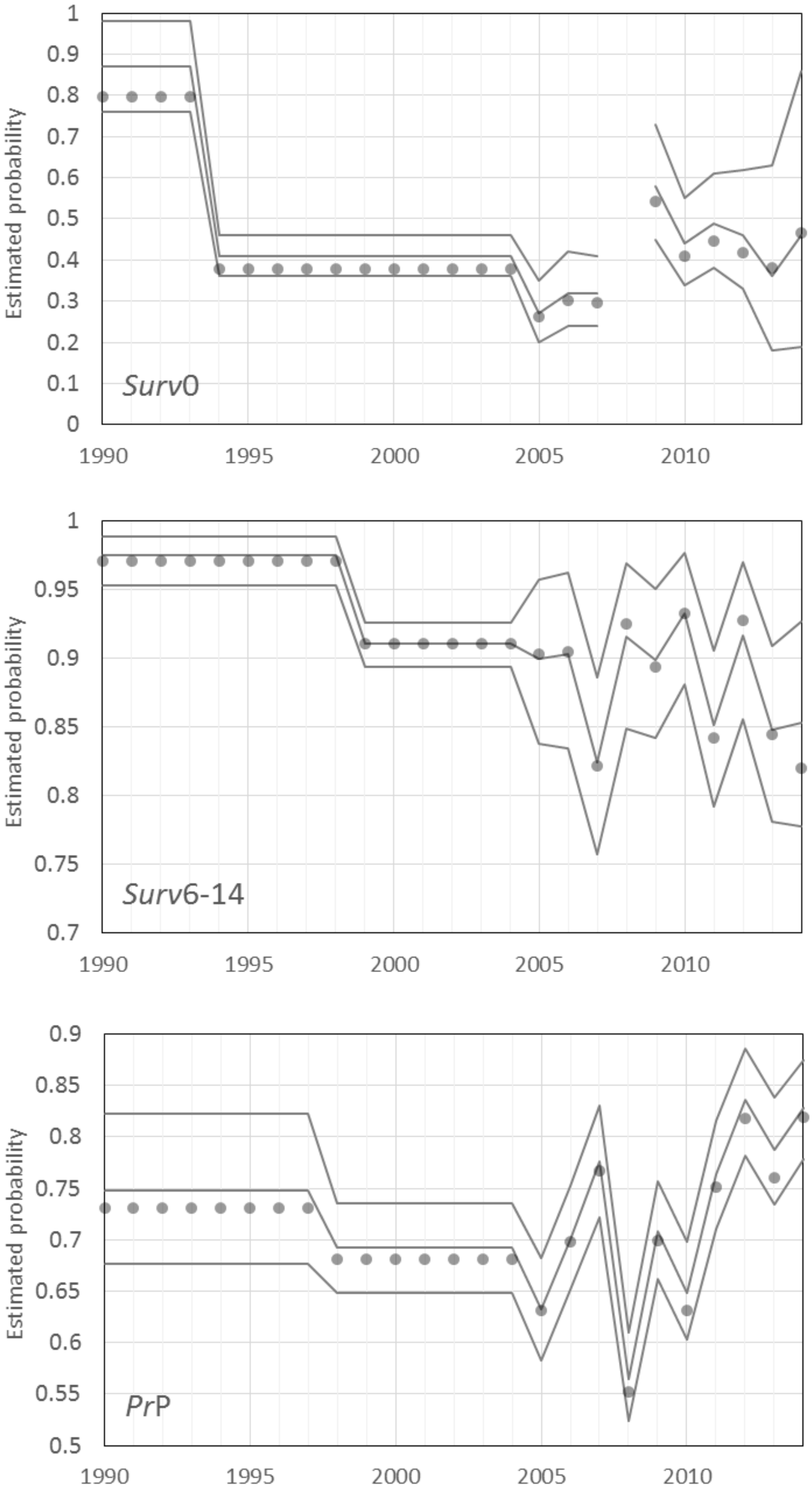

Figure 4.4: Annual estimates of pup survival to age 1 y (top), annual survival at age 6–14 y (middle), and annual probability of pupping (bottom) of female New Zealand sea lions at the Auckland Islands; points are point estimates; lines are median estimates and 95% c.i. (Roberts & Doonan 2016, updated for AEBAR 2018). Note that terminal estimates of very high pupping rates (> 80%) may be implausible; it is likely that annual data collection favouring Sandy Bay rather than Dundas Island, is forcing the model to explain increased pop production as a consequence of increased pupping rate rather than increased female population size at Dundas Island. Collecting mark-recapture data from Dundas Island would address this potential issue, but will face logistical obstacles.

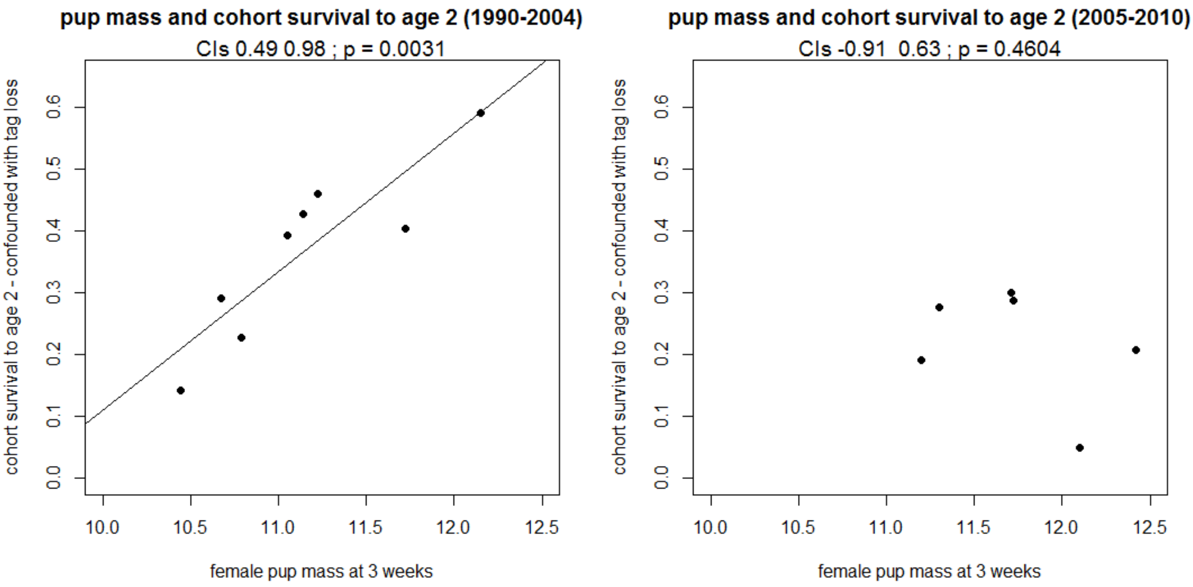

Figure 4.5: Pup mass of females and demographic modelling estimate of cohort survival to age 2; survival estimates confounded with tag loss rate; regression line shown for correlations significant at the 5% level.

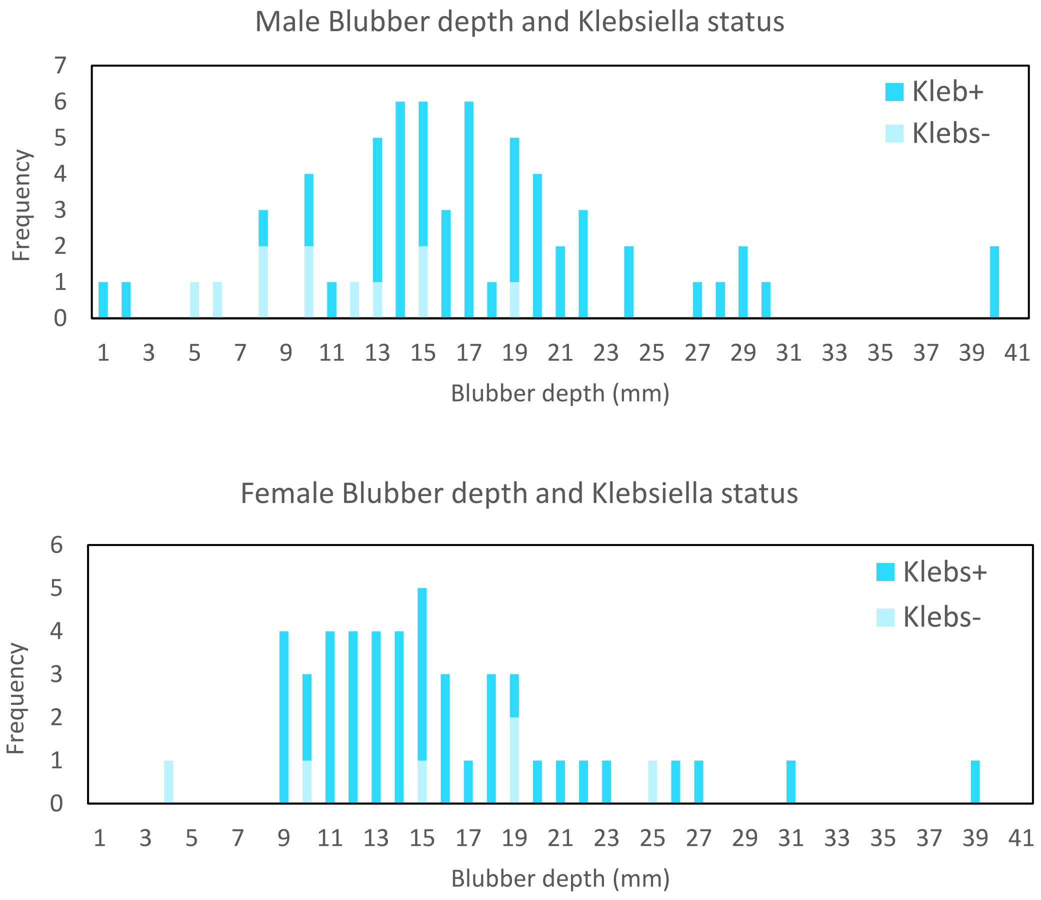

Figure 4.6: Blubber depth of necropsied Auckland Islands sea lion pups for which *Klebsiella pneumoniae* was or was not identified as the cause of death based on histology and/or tissue culture (preliminary data from W. Roe unpublished). All bodies were sampled in February, in the 2006–07 to 2009–10 and 2013–14 field seasons.

### 4.2.9 CONSERVATION THREAT STATUS CLASSIFICATION

Threat classification is an established approach for identifying species at risk of extinction (IUCN 2010). The threat status of New Zealand sea lions has been assessed under two threat classification systems, the International Union for the Conservation of Nature (IUCN) Red List of Threatened Species (IUCN 2010) and the New Zealand Threat Classification System (Townsend et al. 2008).

In 2015, the IUCN updated the Red List status of New Zealand sea lions, listing them as Endangered, on the basis of a projected decline in breeders over three generations (calculated to be 32 years) exceeding a 50% reduction (estimated to be 72%), assuming a linear extrapolation of the observed rate of decline in pup production at the Auckland Islands between 1997–98 and 2008–09 (Chilvers 2015). In 2013, the threat status of New Zealand sea lions was changed from At Risk, Range Restricted[2] to Nationally Critical under criterion C[3] (with a Range Restricted qualifier) based on the same observations of declining population trend at the Auckland Islands (Baker et al. 2016).

In 2019 the New Zealand Threat Classification status for New Zealand sea lions was updated to 'Nationally Vulnerable' (Baker et al. 2019) reflecting that the formally declining population trend at the Auckland Islands has been approximately stable since 2010, and populations are increasing or stable at all other locations.

## 4.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

Reviews of fisheries interactions among pinnipeds globally can be found in Read et al. (2006), Woodley & Lavigne (1991), Katsanevakis (2008) and Moore et al. (2009). Because New Zealand sea lions are endemic to New Zealand, the global understanding of fisheries interactions for this species is outlined under state of knowledge in New Zealand.

## 4.4 STATE OF KNOWLEDGE IN NEW ZEALAND

New Zealand sea lions interact with some trawl fisheries, sometimes resulting in incidental capture and death of the sea lion in the net. Observed trawl fishery interactions are confined to sub-Antarctic waters (Figure 4.7); particularly the two trawl fisheries around the Auckland Islands – the arrow squid fishery (SQU 6T) and the scampi fishery (SCI 6A). Male sea lions are caught in the southern blue whiting fishery near Campbell Island (SBW 6I) and occasional mostly male captures occur in the Stewart-Snares shelf in trawl fisheries targeting mainly arrow squid (SQU 1T; Thompson & Abraham 2010, Thompson et al. 2011, 2013).[4] New Zealand sea lions can forage to depths of 600 m but mainly overlap with trawling at depths of 180–220 m for trawls targeting arrow squid, 250–600 m for trawls targeting spawning southern blue whiting, and 350–550 m for trawls targeting scampi (Tuck 2009, Fisheries New Zealand 2020).

There is seasonal variation in the overlap between New Zealand sea lions and the target species fisheries (Table 4.5) Breeding male sea lions in the Auckland Islands area are ashore between November and January with occasional trips to sea, then migrate away from the area (Robertson et al. 2006). Breeding females are in the Auckland Islands area year-round, coming ashore for up to 10 days to give birth during December and January and then alternately foraging at sea (for about 2 days) and suckling their pup ashore (about 1.5 days; Chilvers et al. 2005b). The SQU 6T fishery currently operates between December and June, peaking between February and May, whereas the SQU 1T fishery operates between December and May, peaking between January and April, before the squid spawn. The SBW 6I fishery operates in August and September, peaking in the latter month, when the fish aggregate to spawn. The SCI 6A fishery typically operates between May and October. Table 4.5: Monthly distribution of New Zealand sea lion activity and seasonal activity of trawl fisheries in which captures may occur.

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[2] A taxon is listed as 'Range Restricted' if it is confined to specific substrates, habitats, or geographic areas of less than 1000 km$^2$ (100 000 ha); this is assessed by taking into account the area of occupied habitat of all subpopulations (Townsend et al. 2008).

[3] A taxon is listed as 'Nationally Critical' under criterion C if the population (irrespective of size or number of subpopulations) has a very high (rate of) ongoing or predicted decline; greater than 70% over 10 years or three generations, whichever is longer (Townsend et al. 2008).

[4] See Fisheries plenary report (Fisheries New Zealand 2020) https://www.mpi.govt.nz/news-and-resources/science-and-research/fisheries-research.

Table 4.5: Monthly distribution of New Zealand sea lion activity and seasonal activity of trawl fisheries in which captures may occur.

| **New Zealand sea lions** | **Sep** | **Oct** | **Nov** | **Dec** | **Jan** | **Feb** | **Mar** | **Apr** | **May** | **Jun** | **Jul** | **Aug** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Breeding males | Dispersed at sea or at haulouts | | At breeding colony | | | Dispersed at sea or at haulouts | | | | | | |
| Breeding females | At sea | | | At breeding colony | | At breeding colony and at-sea foraging and suckling | | | | | | |
| New pups | | | | At breeding colony | | | | | | | | |
| Non-breeders | Dispersed at sea, at haulouts, or at breeding colony periphery | | | | | | | | | | | |
| **Major fisheries** | **Sep** | **Oct** | **Nov** | **Dec** | **Jan** | **Feb** | **Mar** | **Apr** | **May** | **Jun** | **Jul** | **Aug** |
| Squid | | | | Stewart-Snares shelf | | Auckland Islands Shelf and Stewart-Snares shelf | | | | | | |
| Southern blue whiting | Pukaki Rise and Campbell Rise | | | | | | | | | | | Bounty Islands |
| Scampi | Auckland Islands | | | | | | | | Auckland Islands | | | |

### 4.4.1 DIRECT ESTIMATION OF FISHERIES CAPTURES

Incidental captures of New Zealand sea lions are recorded by fisheries observers and used to estimate total capture rates including in unobserved fishing events.

From 2007 to 2017, in fisheries with low numbers of observed captures, total captures were estimated using simple ratio estimates; these included Auckland Islands scampi (SCI 6A) fishery, other Auckland Islands trawl fisheries, and the Stewart-Snares shelf fisheries (Thompson et al. 2013, Abraham & Berkenbusch 2017). Observed annual captures by target fishery are shown in Tables 4.6 and 4.7. Modeled total capture estimates for the Auckland Islands trawl fisheries have subsequently been replaced by spatially explicit estimation methods (Large et al. 2019; see below).

Model estimates by Abraham & Berkenbush (2017) for the Campbell Island and Snares-Stewart shelf fisheries are given in Table 4.6. Observed and estimated New Zealand sea lion captures and capture rates in the SBW 6I fishery have been highly variable. Following the 2012–13 season in which 21 male sea lion captures were observed in a very short period (17 dead and 4 released alive), the fishing industry took immediate action in consultation with the Crown to mitigate sea lion mortalities including 100% use of Sea Lion Exclusion Devices (SLEDs; see below). Since that time 100% of tows have been observed (Table 4.6); annual captures have ranged from 2 to 6 sea lions (annual average under 3).

For the SQU 6T and SBW 6I fisheries, in which the majority of historical captures have been observed, early models suggested that the rate at which sea lions interacted with trawl nets was influenced by a number of factors, including year, distance from the colony, tow duration, time of day, and change of tow direction (Smith & Baird 2005). Subsequently, Thompson et al. (2013) and Abraham & Berkenbusch (2017) applied Bayesian models using these and other categorical covariates to estimate total capture levels thereafter, with greater than 85% observer coverage since the 2012–13 fishing year.

For fisheries with observer coverage approaching 100%, statistical captures estimation that simply scales up from the observed to unobserved fishing effort became largely unnecessary. However, early in the same period during which observer coverage was increasing (i.e., 2001–02 to 2007–08), the SQU 6T fishery also transitioned to widespread adoption of SLEDs, a mitigation device designed to allow sea lions entering the trawl net to exit via the SLED and survive. Unsurprisingly, following the introduction of SLEDs to the SQU 6T fishery in 2001–02, both the observed and estimated numbers of New Zealand sea lion captures declined (Table 4.8 and 4.9). However since the universal adoption of a single ‘standard’ SLED design in 2006–07, model estimates of *interaction rates* — i.e., the number of sea lions entering the net but potentially exiting again via the SLED — became increasingly uncertain over time, because the interaction rate was confounded in the model with a corresponding and inversely correlated estimate of SLED efficacy, both of which were effectively unobservable. In the most recent such models for Auckland Islands sea lions (Abraham & Berkenbusch 2017), the estimated interaction rates were effectively unbounded, and model estimates in particular years became unstable as new years of data were added. For this reason, from 2017 Fisheries New Zealand discontinued these models for Auckland Islands sea lions and applied a new approach under which *interactions, captures*, and *deaths* are estimated separately, combining an application of the spatial risk assessment (SEFRA) method described in Chapter 3 with a separate means of estimating cryptic mortality (i.e., unobservable deaths) as a function of the observable captures in trawls employing SLEDs. The outcomes of this new body of research are described below.

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Table 4.6: Sea lion captures in Campbell Island trawl fisheries targeting southern blue whiting (SBW) and in Stewart-Snares shelf trawl fisheries targeting squid (SQU 1T) and all other deepwater target species, between 2002–03 and 2017–18 (https://psc.dragonfly.co.nz/2019v1/released/new-zealand-sea-lion/southern-blue-whiting-trawl/all-vessels/eez/2017-18/). Annual fishing effort (total number of tows), observer coverage (percentage of tows observed), number of observed sea lion captures (both dead and alive), observed capture rate (captures per 100 tows), the estimation method used (model or ratio estimate), and the mean number of estimated sea lion captures (with 95% confidence interval, c.i.) (see Thompson et al. 2013 and 2016 for details). Data for subsequent years are provided by Fisheries New Zealand.

| Fishing year | Fishing | Observed captures | | | Estimated captures | | |
|---|---|---|---|---|---|---|---|
| | All effort | % observed | Number | Rate | Method | Mean | 95% c.i. |
| Campbell Island SBW | | | | | | | |
| 2002–03 | 638 | 43 | 0 | 0 | Model | 1 | 0–3 |
| 2003–04 | 740 | 33 | 1 | 0.4 | Model | 3 | 1–9 |
| 2004–05 | 870 | 38 | 2 | 0.6 | Model | 5 | 2–13 |
| 2005–06 | 624 | 35 | 3 | 1.4 | Model | 10 | 3–22 |
| 2006–07* | 630 | 36 | 3 | 1.3 | Model | 15 | 6–30 |
| 2007–08 | 818 | 41 | 5 | 1.5 | Model | 8 | 5–14 |
| 2008–09 | 1188 | 25 | 0 | 0 | Model | 1 | 0–7 |
| 2009–10 | 1114 | 36 | 11 | 2.8 | Model | 24 | 15–37 |
| 2010–11 | 1171 | 37 | 6 | 1.4 | Model | 15 | 8–25 |
| 2011–12 | 951 | 70 | 0 | 0 | Model | 1 | 0–4 |
| 2012–13 | 790 | 100 | 21 | 2.7 | N/A | 21 | 21–21 |
| 2013–14 | 809 | 100 | 2 | 0.3 | N/A | 2 | 2–2 |
| 2014–15 | 677 | 99 | 6 | 0.9 | N/A | 6 | 6–6 |
| 2015–16 | 442 | 100 | 3 | 0.7 | N/A | 3 | 3–3 |
| 2016–17 | 539 | 100 | 0 | 0 | N/A | 0 | 0 |
| 2017–18 | 455 | 100 | 2 | 0.4 | N/A | 2 | 2–2 |
| 2018–19 | 424 | 100 | 0 | 0 | N/A | 0 | 0 |
| Stewart-Snares shelf | | | | | | | |
| 2002–03 | 7297 | 15 | 0 | 0.00 | Ratio | 3 | 0–7 |
| 2003–04 | 7493 | 15 | 1 | 0.09 | Ratio | 4 | 1–8 |
| 2004–05 | 8419 | 21 | 3 | 0.17 | Ratio | 7 | 3–11 |
| 2005–06 | 7342 | 15 | 1 | 0.09 | Ratio | 4 | 1–8 |
| 2006–07 | 6246 | 20 | 1 | 0.08 | Ratio | 3 | 1–6 |
| 2007–08 | 5040 | 29 | 1 | 0.07 | Ratio | 3 | 1–6 |
| 2008–09 | 4348 | 29 | 0 | 0.00 | Ratio | 1 | 0–4 |
| 2009–10 | 4960 | 35 | 1 | 0.06 | Ratio | 2 | 1–5 |
| 2010–11 | 4432 | 29 | 0 | 0.00 | Ratio | 1 | 0–4 |
| 2011–12 | 4536 | 36 | 1 | 0.06 | Ratio | 2 | 1–4 |
| 2012–13 | 4301 | 70 | 1 | 0.03 | Ratio | 2 | 1–4 |
| 2013–14 | 4185 | 59 | 0 | 0.00 | Ratio | 1 | 0–3 |
| 2014–15 | 4154 | 57 | 1 | 0.04 | Ratio | 1 | 0–3 |
| 2015–16 | 2978 | 66 | 1 | 0.05 | | | |
| 2016–17 | 3603 | 54 | 0 | 0.00 | | | |
| 2017–18 | 3797 | 75 | 1 | 0.04 | | | |

* Standardised SLED design introduced in this year.

Relatively high observed capture rates of sea lions in the SQU 6T fishery before 2002, with moderate fishing observer coverage and highly variable total effort, suggested that substantial numbers of captures may be occurring. Observer coverage levels increased substantially in the SQU 6T fishery in the years since the 2002–03 season. Observer coverage ranged from 28–45% between the 2002–03 and 2011–12 fishing seasons and achieved high

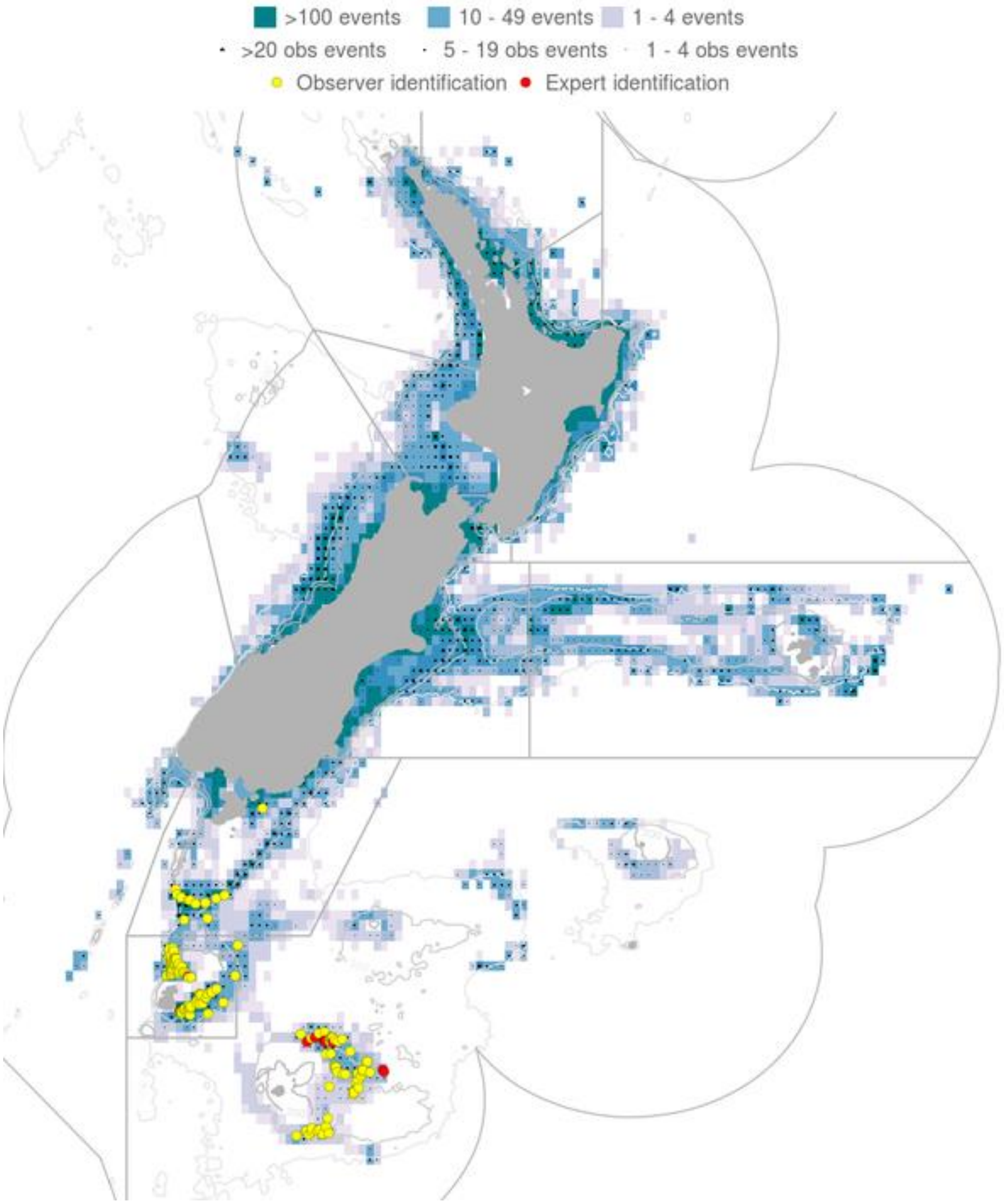

Figure 4.7: Spatial distribution of trawl fishing effort and observed New Zealand sea lion captures, 2002–03 to 2017–18 (https://psc.dragonfly.co.nz/2019v1/released/new-zealand-sea-lion/trawl/all-vessels/eez/2002-03-2017-18/). Fishing effort density is mapped into 0.2-degree cells, in blue and green. The corresponding level of fisheries observer coverage is indicated by the superimposed black dots; observed captures are indicated in yellow or red.

Table 4.7: Sea lion captures in Auckland Islands trawl fisheries, for trawls targeting scampi (SCI) and trawls targeting all species other than scampi and squid, from 1992–93 to 2016–17 (from Large et al. 2019). Data for subsequent years are provided by Fisheries New Zealand. Columns denote annual fishing effort (total number of tows), observer coverage (percentage of tows observed), and number of observed sea lion captures (combined for male and female sea lions, including both live and dead captures). Corresponding estimates of total fisheries deaths are shown in Figure 4.11 and Table 4.9.

| Fishing year | SCI trawl | | | Other trawl | | |
|---|---|---|---|---|---|---|
| | All effort | Observed (%) | Captures | All effort | Observed (%) | Captures |
| 1992–93 | 835 | 18 | 3 | 195 | 18 | 0 |
| 1993–94 | 1 314 | 21 | 0 | 308 | 7 | 0 |
| 1994–95 | 1 349 | 4 | 0 | 492 | 7 | 0 |
| 1995–96 | 1 312 | 5 | 2 | 411 | 6 | 1 |
| 1996–97 | 1 227 | 16 | 1 | 296 | 4 | 0 |
| 1997–98 | 1 109 | 12 | 0 | 688 | 17 | 0 |
| 1998–99 | 1 255 | 2 | 0 | 525 | 10 | 0 |
| 1999–00 | 1 383 | 5 | 0 | 751 | 13 | 0 |
| 2000–01 | 1 419 | 6 | 4 | 577 | 7 | 0 |
| 2001–02 | 1 603 | 10 | 0 | 590 | 4 | 0 |
| 2002–03 | 1 351 | 11 | 0 | 543 | 13 | 1 |
| 2003–04 | 1 363 | 12 | 3 | 289 | 17 | 0 |
| 2004–05 | 1 275 | 0 | – | 170 | 7 | 0 |
| 2005–06 | 1 331 | 9 | 1 | 39 | 15 | 0 |
| 2006–07 | 1 328 | 8 | 1 | 38 | 5 | 0 |
| 2007–08 | 1 327 | 7 | 0 | 147 | 45 | 0 |
| 2008–09 | 1 457 | 4 | 1 | 121 | 50 | 0 |
| 2009–10 | 940 | 10 | 0 | 77 | 68 | 0 |
| 2010–11 | 1 401 | 15 | 0 | 131 | 37 | 0 |
| 2011–12 | 1 247 | 10 | 0 | 57 | 30 | 0 |
| 2012–13 | 1 093 | 12 | 0 | 60 | 43 | 0 |
| 2013–14 | 850 | 6 | 0 | 203 | 23 | 0 |
| 2014–15 | 548 | 0 | – | 224 | 31 | 0 |
| 2015–16 | 1 414 | 5 | 0 | 140 | 26 | 0 |
| 2016–17 | 1 677 | 21 | 0 | 170 | 51 | 0 |
| 2017–18 | 1 728 | 17 | 2 | 146 | 57 | 0 |

Table 4.8: Sea lion captures in Auckland Islands trawl fisheries targeting squid, shown separately for bottom trawl and midwater trawl gear configurations, from 1992–93 to 2016–17 (from Large et al. 2019). Data for subsequent years are provided by Fisheries New Zealand. Columns denote annual fishing effort (total number of tows), observer coverage (percentage of tows observed), and number of observed sea lion captures (separately for female and male sea lions, including both live and dead captures. Corresponding estimates of total fisheries deaths are shown in Figure 4.11 and Table 4.9. (continued next page)

| Fishing year | Bottom trawl effort | | | | Midwater trawl effort | | | |
|---|---|---|---|---|---|---|---|---|
| | All effort | Observed (%) | Female captures | Male captures | All effort | Observed (%) | Female captures | Male captures |
| 1992–93 | 86 | 10 | 0 | 0 | 568 | 33 | 3 | 2 |
| 1993–94 | 0 | – | 2 | 1 | 3 226 | 7 | 0 | 1 |
| 1994–95 | 0 | – | 1 | 2 | 2 633 | 7 | 3 | 2 |
| 1995–96 | 721 | 0 | 0 | 0 | 3 747 | 15 | 10 | 3 |
| 1996–97 | 0 | – | 2 | 7 | 2 177 | 25 | 7 | 12 |
| 1997–98 | 242 | 19 | 2 | 2 | 1 219 | 24 | 2 | 9 |
| 1998–99 | 89 | 33 | 1 | 0 | 313 | 41 | 3 | 1 |
| 1999–00 | 455 | 15 | 1 | 0 | 751 | 50 | 12 | 12 |
| 2000–01 | 173 | 99 | 6 | 4 | 410 | 99 | 16 | 13 |
| 2001–02* | 498 | 21 | 2 | 0 | 1 149 | 40 | 12 | 7 |
| 2002–03* | 738 | 34 | 2 | 1 | 728 | 23 | 5 | 3 |
| 2003–04* | 1 452 | 17 | 3 | 1 | 1 142 | 47 | 11 | 1 |
| 2004–05* | 1 375 | 21 | 5 | 2 | 1 318 | 39 | 0 | 2 |
| 2005–06* | 1 905 | 13 | 3 | 0 | 554 | 55 | 7 | 0 |
| 2006–07* | 732 | 43 | 2 | 1 | 585 | 38 | 4 | 0 |
| 2007–08* | 634 | 43 | 2 | 2 | 631 | 50 | 1 | 0 |

| Fishing year | Bottom trawl effort | | | | Midwater trawl effort | | | |
|---|---|---|---|---|---|---|---|---|
| | All effort | Observed (%) | Female captures | Male captures | All effort | Observed (%) | Female captures | Male captures |
| 2008–09+ | 1 068 | 34 | 1 | 1 | 857 | 46 | 0 | 0 |
| 2009–10+ | 1 026 | 23 | 2 | 0 | 162 | 41 | 1 | 0 |
| 2010–11+ | 1 218 | 30 | 0 | 0 | 365 | 49 | 0 | 0 |
| 2011–12+ | 973 | 34 | 0 | 0 | 308 | 78 | 0 | 0 |
| 2012–13+ | 813 | 83 | 3 | 0 | 214 | 100 | 0 | 0 |
| 2013–14+ | 477 | 83 | 2 | 0 | 260 | 87 | 0 | 0 |
| 2014–15+ | 328 | 92 | 0 | 0 | 305 | 84 | 1 | 0 |
| 2015–16+ | 822 | 87 | 0 | 0 | 543 | 100 | 0 | 0 |
| 2016–17+ | 1 090 | 67 | 2 | 0 | 204 | 78 | 1 | 0 |
| 2017–18+ | 987 | 88 | 2 | 0 | 143 | 100 | 0 | 0 |
| 2018–19+ | 712 | 96 | 3 | 4 | 94 | 88 | 0 | 0 |

* denotes years in which SLEDS were deployed on a variable proportion of trawls, in the absence of a standard design or systematic inspection and audit programme.

+ denotes years in which SLEDs were deployed universally on all trawls, with a standard design and a systematic inspection and audit programme.

## 4.4.2 SPATIAL FISHERIES RISK ASSESSMENT FOR AUCKLAND ISLANDS SEA LIONS

The widespread introduction of SLEDs in the SQU 6T trawl fishery created the need for a new modelling approach in which *interactions, captures,* and *deaths, including cryptic deaths,* can be estimated separately, and under which estimation of the interaction rate is not confounded by the SLED efficacy rate. These terms are defined clearly here to avoid confusion, noting that before the adoption of the new modelling approach in 2019, the same terms may have been applied in a less consistent way.

*Captures* are sea lions captured in nets and brought on deck (both dead and alive). Captures necessarily exclude the animals that exit trawls through the SLED, as well as bodies that are recovered in a decomposed state hence presumed to be already dead at the time that the body entered the net (Smith & Baird 2007b, Thompson & Abraham 2010, Thompson et al. 2013).

*Interactions* in the SQU 6T fishery are defined as the number of sea lions that enter the net alive and would have been captured if no SLED had been used. Until 2017 interactions were estimated using a statistical model fitting to observed capture rates both before and after the deployment of SLEDs, with an additional term to approximate the presumed level of 'SLED efficacy', i.e., the proportion of interactions in which the sea lion exits via the SLED and survives (Thompson et al. 2013). For trawl fisheries that do not deploy SLEDs, the number of estimated interactions is equivalent to the number of estimated captures.

*Deaths* include both observable captures (excluding animals released alive and presumed to survive) and also *cryptic deaths*, i.e., animals which are not recovered on board the vessel or otherwise observable (i.e., 'captured') even in the presence of a fisheries observer, but are nonetheless expected to die as a direct consequence of their interaction with the fishing gear.

Prior to the introduction of SLEDs there was no feasible mechanism by which sea lion bodies could be accounted for if lost or unable to be observed, such that *interactions* = *captures* and *captures* $\geq$ *deaths* (i.e., *cryptic deaths* = 0). After the successful introduction of SLEDs, the modelling approach used previously became increasingly ill-suited to estimating sea lion deaths due to uncertainties about the rate at which sea lions were exiting via the SLED, and the potential for cryptic mortality. From 2019 an adaptation of the SEFRA approach outlined in Chapter 3, was used instead, in which each of the critical rates are estimated empirically. See Large et al. (2019) for a full description of the spatial risk modelling for Auckland Islands sea lions; key outputs of this work are summarised below.

### 4.4.2.1 SPATIAL FORAGING DISTRIBUTION MODELLING

Satellite telemetry data indicative of spatial foraging patterns for Auckland Islands sea lions were compiled and analysed to predict the most likely foraging track per trip (i.e., removing implausible location outliers). Tracks were then used to estimate the spatial density of female sea lions (estimated separately for adults and juveniles). Due to the high density of the available telemetry data, and the incomplete spatial coverage of prey availability information, the best fitting spatial models used simple geographic covariates (latitude, longitude, depth, and distance to colony) rather than true habitat variables. Groomed satellite telemetry data and the resulting combined density layer, scaled for the relative abundance of adults and juveniles, are shown in Figure 4.8.

Note that to the extent that the three different breeding colonies (Sandy Bay, Dundas Island, and Figure of Eight Island) may exhibit distinct spatial foraging patterns, there may be spatial biases in the estimated spatial density used to inform the risk assessment, reflecting that the majority of the data were collected at Sandy Bay. Furthermore, all available data were collected in summer; the seasonal bias will not affect estimation of fisheries risk in the squid fishery (which occurs in summer) but may create additional uncertainty with respect to fishing in other seasons (e.g., scampi trawl fisheries). Additional tracking studies are planned to address these potential imbalances.

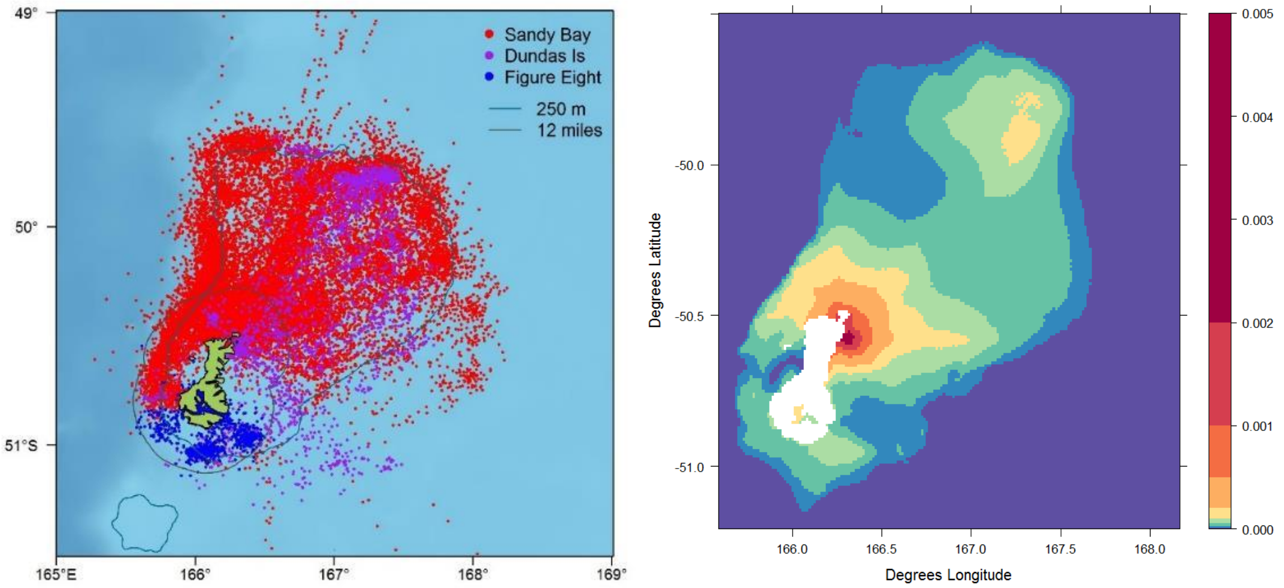

Figure 4.8: Groomed and filtered fix locations for female sea lions tagged at three Auckland Islands breeding colonies (left) and estimated spatial density (on a relative scale; all cell values sum to 1) of all female Auckland Islands sea lions (age 2+), as used in spatial risk models (right). Reproduced from Large et al. (2019).

### 4.4.2.2 SEA LION CATCHABILITY IN COMMERCIAL TRAWL FISHERIES

Under the SEFRA method, encounters between sea lions and fishing effort are proportional to the spatial overlap between the sea lion distribution and the distribution of fishing effort. Catchability is the probability of capture per encounter.

Catchability was estimated separately in eight different trawl fishery groups, as follows. First, fishery groups were divided by target fishery, i.e., squid target fisheries vs. scampi target fisheries vs. ‘other deepwater trawl’. Next, within the squid fishery, catchability was estimated separately for ‘bottom trawl’ vs. ‘midwater trawl’ gear configurations (noting that both gear configurations are actually deployed in contact with the sea floor, but the length and headline height of the nets differs between them). Finally each squid fishery gear type was divided into three categories relating to the deployment of SLEDs as follows:

- no SLED deployed (all effort prior to the 2001 season) and a declining proportion of effort thereafter

- non-standard SLED: SLEDs deployed during the years 2001–2007, during which the design of the SLED had not been standardised and there was no systematic audit to ensure proper deployment

- standard SLED: from 2008 onward, all fishing effort used a standardised SLED design subject to verification under a systematic inspection process (Cleal et al. 2007)

Sea lion catchability in these eight fishery groups is shown in Figure 4.9. This figure indicates that catchability may be highest in ‘other trawl’ fisheries (e.g., hoki trawl), but this result is highly uncertain and has almost no contribution to actual risk because spatial overlap is very low (i.e., there is almost no fishing effort in this category occurring near the

Auckland Islands). Of the fisheries that do overlap with Auckland Islands sea lions, catchability is estimated to be highest in scampi trawls.

Comparing catchability estimates among squid fishery groups reveals important patterns. In both the midwater and bottom trawl gear configurations, the effectiveness of SLEDs at reducing catchability is clear, but this effect was only realised after SLED designs were standardised and audited, from 2008. When catchability in midwater trawls vs. bottom trawls is compared, it appears that without the use of SLEDs, sea lions are more likely to be captured in midwater gear, but with SLEDs, captures are more likely in bottom trawl gear. These results imply that in a given encounter with fishing effort, a sea lion is more likely to enter a midwater net than a bottom trawl net, but is also more likely to exit successfully from a midwater net via the SLED. This may be related to the higher headline height of midwater nets relative to bottom trawl nets; note however that these indications are uncertain (the confidence intervals overlap).

#### 4.4.2.3 ESTIMATED CAPTURES, DEATHS, AND POPULATION-LEVEL RISK OVER TIME

Because spatially resolved fishing effort data are available from 1993, by applying the estimated catchabilities in Figure 4.9 it is possible to estimate historical changes in fisheries captures over time, including the effect of changing effort levels, changing spatial fishing effort patterns, and changing sea lion population sizes. Cryptic mortality in trawls employing SLEDs is estimated separately by Meyer (2019; see below); risk reflects fisheries deaths as a proportion of population size, with an implied population impact limit (which is a policy decision). Figures 4.10a and 4.10b. show that estimated sea lion deaths in the squid fishery peaked in the early to mid-1990s, declined to a low in 1999 reflecting greatly reduced fishing effort levels, increased again as effort levels increased to a lower peak in 2006, then declined dramatically reflecting the universal adoption of standardised SLEDs, which reduced sea lion catchability from the 2009 season onward (Large et al. 2019). Cumulative impacts across all trawl fisheries are shown in Figure 4.11 and Table 4.9. Note that the analysis does not include the include the 1980s period of relatively high squid fishery effort, when SLEDs were not used, and annual mortalities were likely to be high relative to the following, assessed period (Large et al. 2019).

These figures suggest that since the universal adoption of standardised SLEDs by the squid fishery in 2008, scampi target fisheries rather than squid target fisheries may now be responsible for the largest proportion of commercial fisheries risk to Auckland Islands sea lions; note however that this conclusion is uncertain due to possible spatial and seasonal biases in the spatial data informing this model.

Table 4.9: Estimated deaths of female Auckland Islands sea lions in trawl fisheries targeting squid, scampi, and other target species, from 1992–93 to 2016–17, from the SEFRA fisheries risk model by Large et al. (2019). Squid trawl fishery estimates combine both the midwater and bottom trawl fishery groups and include cryptic mortality in trawls utilising SLEDs, as estimated by Meyer (2019).

| Fishing year | Squid trawl | | Scampi trawl | | Other trawl | |
|---|---|---|---|---|---|---|
| | Median | 95% c.i. | Median | 95% c.i. | Median | 95% c.i. |
| 1992–93 | 10 | 5–16 | 5 | 1–10 | 0 | 0–2 |
| 1993–94 | 82 | 61–108 | 12 | 4–21 | 1 | 0–4 |
| 1994–95 | 74 | 54–97 | 9 | 4–17 | 1 | 0–3 |
| 1995–96 | 83 | 62–108 | 8 | 3–16 | 1 | 0–6 |
| 1996–97 | 51 | 36–70 | 6 | 2–12 | 1 | 0–5 |
| 1997–98 | 28 | 18–39 | 7 | 2–13 | 2 | 0–8 |
| 1998–99 | 6 | 3–12 | 8 | 3–15 | 1 | 0–4 |
| 1999–00 | 19 | 12–28 | 6 | 2–11 | 1 | 0–5 |
| 2000–01 | 8 | 4–14 | 6 | 2–12 | 0 | 0–3 |
| 2001–02* | 23 | 14–32 | 6 | 2–12 | 1 | 0–5 |
| 2002–03* | 22 | 14–32 | 5 | 1–10 | 1 | 0–5 |
| 2003–04* | 31 | 21–44 | 7 | 2–13 | 0 | 0–1 |
| 2004–05* | 37 | 25–51 | 8 | 3–15 | 0 | 0–1 |
| 2005–06* | 35 | 22–50 | 6 | 2–12 | 0 | 0–1 |
| 2006–07* | 17 | 10–25 | 6 | 2–12 | 0 | 0–0 |
| 2007–08* | 17 | 10–26 | 5 | 1–10 | 0 | 0–1 |
| 2008–09+ | 5 | 2–11 | 7 | 2–13 | 0 | 0–1 |

| Fishing year | Squid trawl | | Scampi trawl | | Other trawl | |
|---|---|---|---|---|---|---|
| | Median | 95% c.i. | Median | 95% c.i. | Median | 95% c.i. |
| 2009–10+ | 4 | 1–8 | 2 | 0–6 | 0 | 0–1 |
| 2010–11+ | 4 | 1–9 | 5 | 12–0 | 0 | 0–1 |
| 2011–12+ | 3 | 0–6 | 3 | 1–8 | 0 | 0–1 |
| 2012–13+ | 3 | 0–6 | 3 | 0–7 | 0 | 0–1 |
| 2013–14+ | 1 | 0–4 | 2 | 0–6 | 0 | 0–1 |
| 2014–15+ | 1 | 0–4 | 1 | 0–4 | 0 | 0–2 |
| 2015–16+ | 2 | 0–6 | 3 | 1–8 | 0 | 0–1 |
| 2016–17+ | 3 | 1–7 | 5 | 1–9 | 0 | 0–2 |

* denotes years in which SLEDS were deployed on a variable proportion of squid target trawls, in the absence of a standard design or systematic inspection and audit programme.

\+ denotes years in which SLEDs were deployed universally on all squid target trawls, with a standard design and a systematic inspection and audit programme.

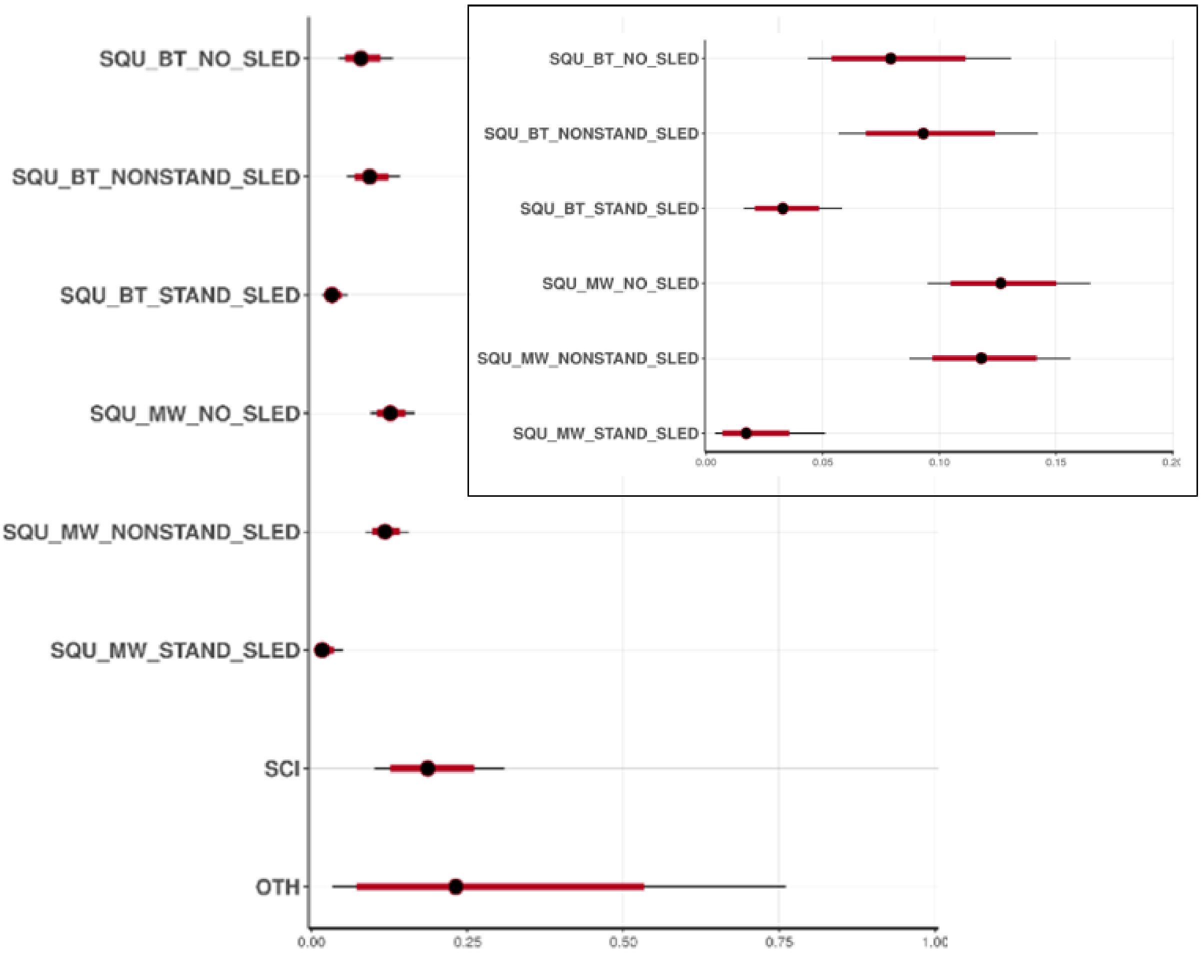

Figure 4.9: Estimated catchability of female New Zealand sea lions in commercial trawl fishery groups: "SQU_BT_NO_SLED" = bottom trawls targeting southern arrow squid without a sea lion exclusion device (SLED), "SQU_BT_NONSTAND_SLED" = bottom trawls targeting squid with a SLED in the period before SLED designs were standardised and universally audited; "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 rescaled to make outputs easier to read. Reproduced from Large et al. (2019).

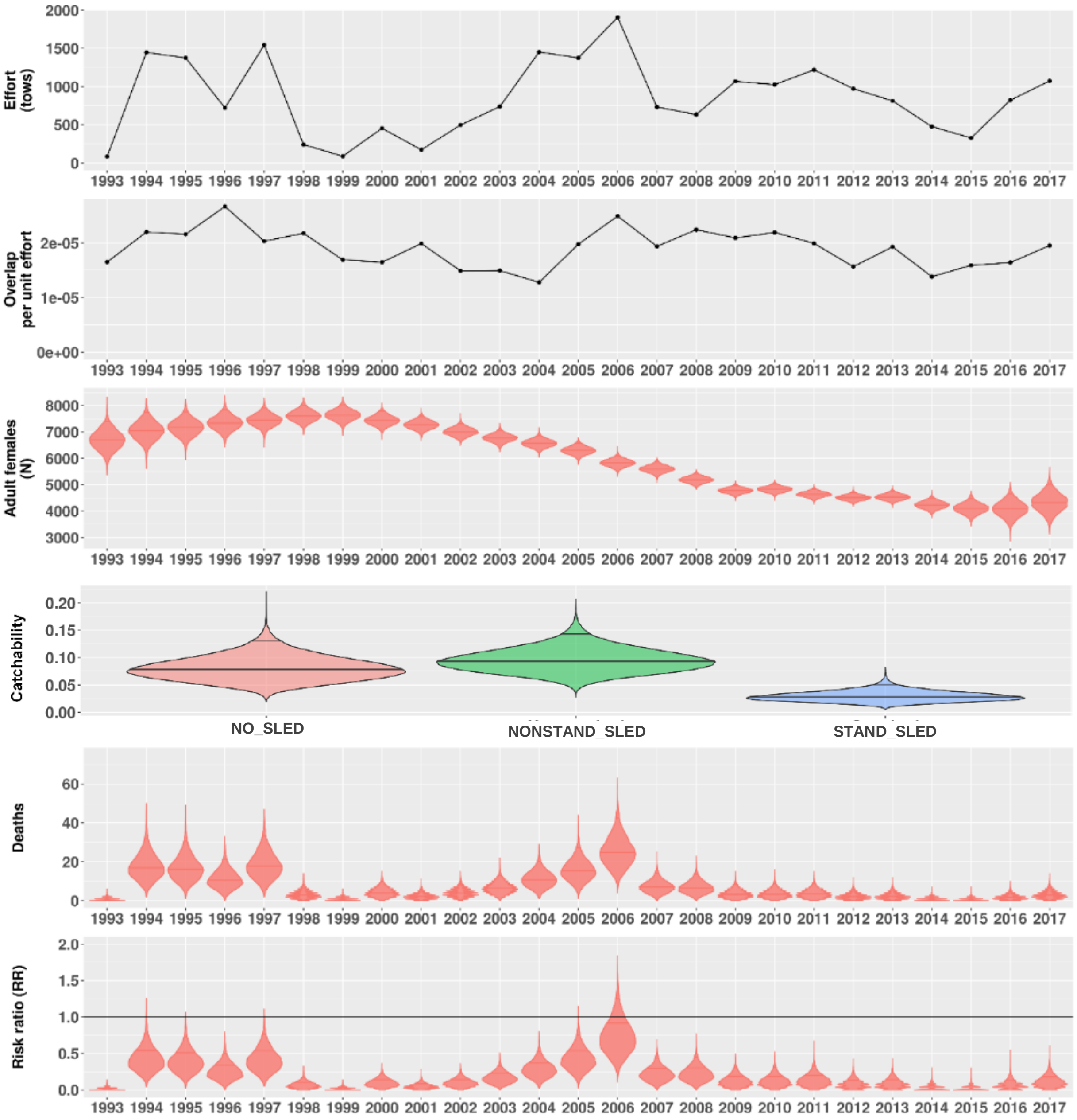

Figure 4.10a: Time series spatial risk model outputs for female Auckland Islands sea lions in the ‘bottom trawl’ fishery group targeting southern arrow squid (SQUBT): total effort; spatial overlap per unit effort; population size (females only); catchability; annual deaths (females only); and risk ratio. Note that the risk ratio in this model run assumed a calibration coefficient ($\phi$) of 0.1. (Large et al. 2019).

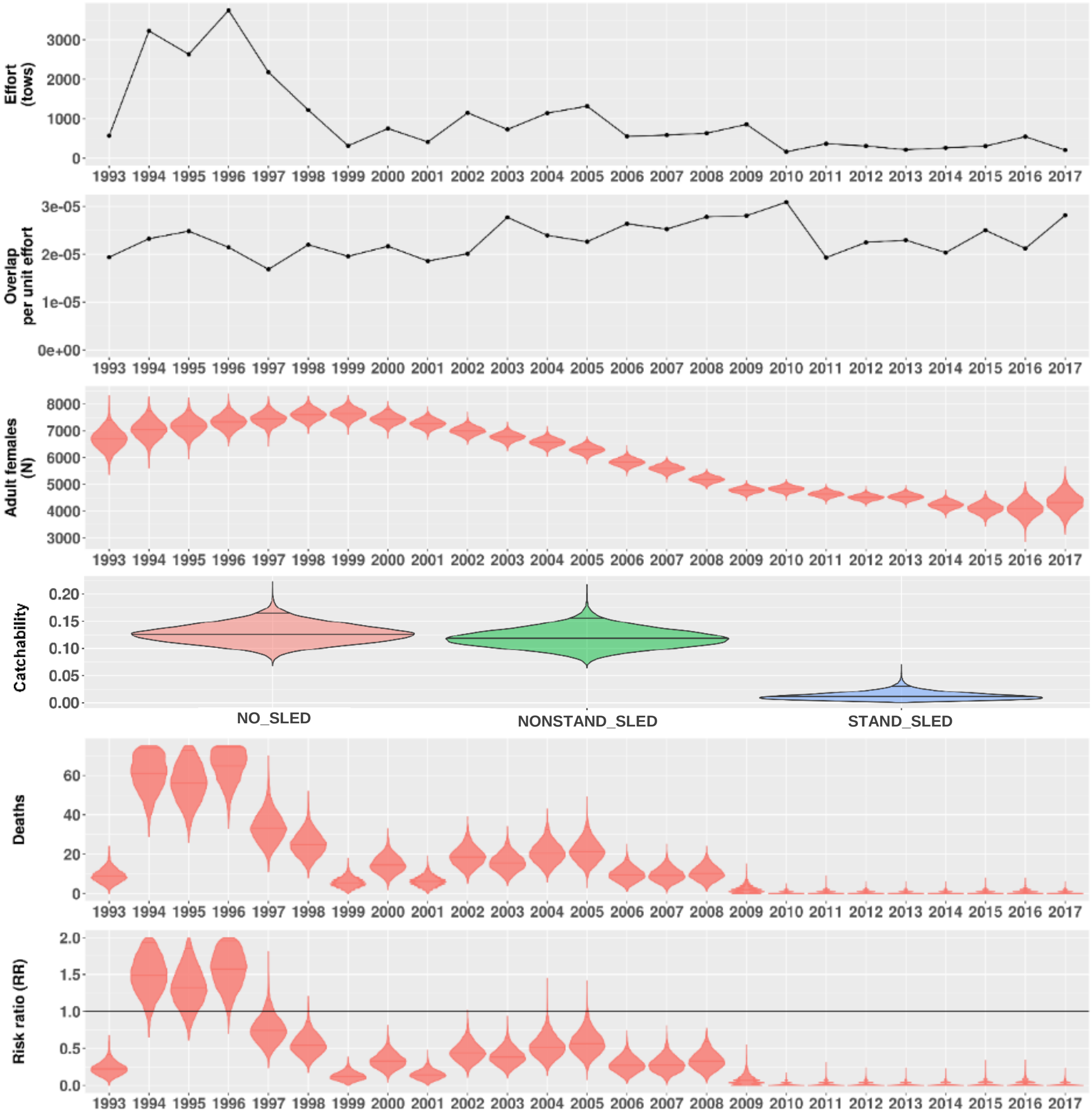

Figure 4.10b: Time series spatial risk model outputs for female Auckland Islands sea lions in the 'midwater trawl' fishery group targeting southern arrow squid (SQUBT): total effort; spatial overlap per unit effort; population size (females only); catchability; annual deaths (females only); and risk ratio. Note that the risk ratio in this model run assumed a calibration coefficient ($\phi$) of 0.1. (Large et al. 2019).

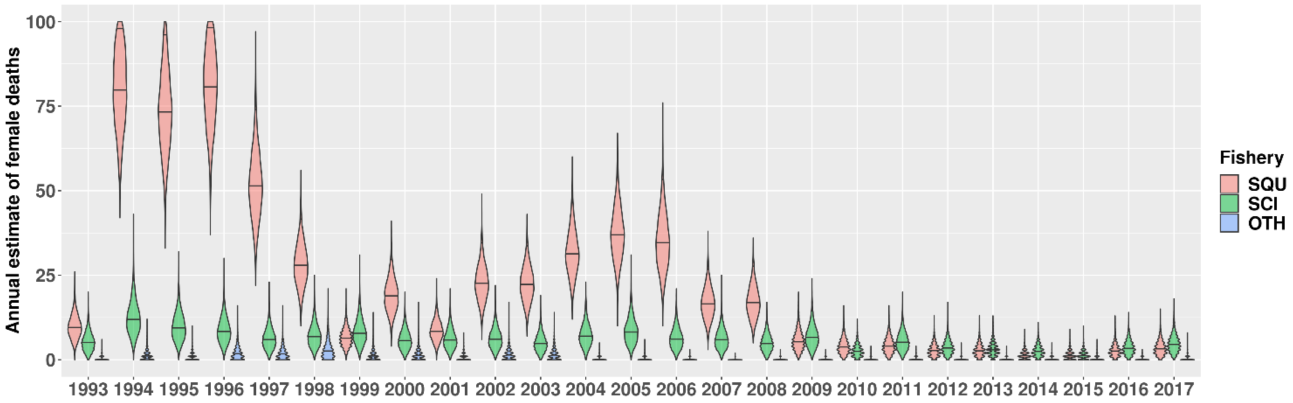

Figure 4.11: Estimated annual deaths of female Auckland Islands sea lions in commercial trawl fisheries from 1992–93 to 2016–17, aggregated by target species: "SQU" = southern arrow squid, "SCI" = scampi, "OTH" = trawls targeting all other species. (Large et al. 2019).

### 4.4.3 SEA LION EXCLUSION DEVICE (SLED) DEVELOPMENT AND USE

In 2004, the Minister of Fisheries requested that Squid Fishery Management Company, government agencies and other interested stakeholders work collaboratively to develop a plan of action to determine SLED efficacy. In response, an independently chaired working group (the SLED Working Group) was established to develop an action plan to determine the efficacy of SLEDs, with a particular focus on the survivability of New Zealand sea lions that exit the nets via the exit hole in the SLED. The group undertook a number of initiatives, most notably the standardisation of SLED specifications (including grid spacing) across the fleet (DOC CSP project MIT 2004/05; Cleal et al. 2007) and the establishment of an underwater video monitoring programme to help understand the fate of New Zealand sea lions that exit the net via the SLED. White light and infra-red illuminators were tested. Sea lions were observed outside the net on a number of occasions, but only one fur seal and one New Zealand sea lion were observed exiting the net via the SLED (on tows when white light illumination was used). The footage contributed to understanding of SLED performance, but established that video monitoring was only suitable for tows using midwater gear, because the camera view was often obscured on tows where bottom gear was used (Middleton & Banks 2008; Middleton 2019a). The SLED Working Group was disbanded in early 2010.

Figures 4.12 and 4.13 illustrate the SLED design and the way in which New Zealand sea lions interact with the trawl gear and the SLED itself.

### 4.4.4 CRYPTIC MORTALITY OF SEA LIONS IN TRAWLS WITH SLEDS

SLEDs are effective in allowing most New Zealand sea lions to exit a trawl (see Figure 4.12) but occasionally a sea lion does not exit and is drowned and retained in the net. These are recorded as observed captures. However there remains some uncertainty about the fate of sea lions that are not retained in the net, some of which may nonetheless die as a consequence of the interaction. Interactions that result in unobservable deaths are termed 'cryptic mortality'. Sources of cryptic mortality are best understood by categorising four potential outcomes of a sea lion entering a trawl:

i. exits the net via SLED and survives (survivor);
ii. dies in net and is retained (observable capture);
iii. dies in the net but the body is subsequently lost without being recovered on the vessel ('body non-retention');
iv. exits the SLED but is at the limit of its 'breath hold' and drowns before reaching the surface ('post-escape drowning').

Collectively, points iii and iv constitute cryptic mortality. Previously a fifth potential outcome had been defined, i.e., v) 'exits the net but dies from head injuries sustained during interaction with the SLED' ('mild traumatic brain injury', or MTBI). However upon review of the process by which sea lions interact with trawls with SLEDs it was judged that the effect of MTBI will be to affect the rate at which a sea lion exits (or does not exit) the net, thereby affecting the likelihood of other outcomes (e.g., capture or post-escape drowning). Therefore MTBI does not constitute a separate outcome in itself (see Figure 4.13).

The following section describes research undertaken to estimate various components potentially contributing to cryptic mortality. In 2019 this information was integrated into a Bayesian estimation model to estimate cryptic mortality empirically (Meyer 2019), the results of which are summarised separately below.

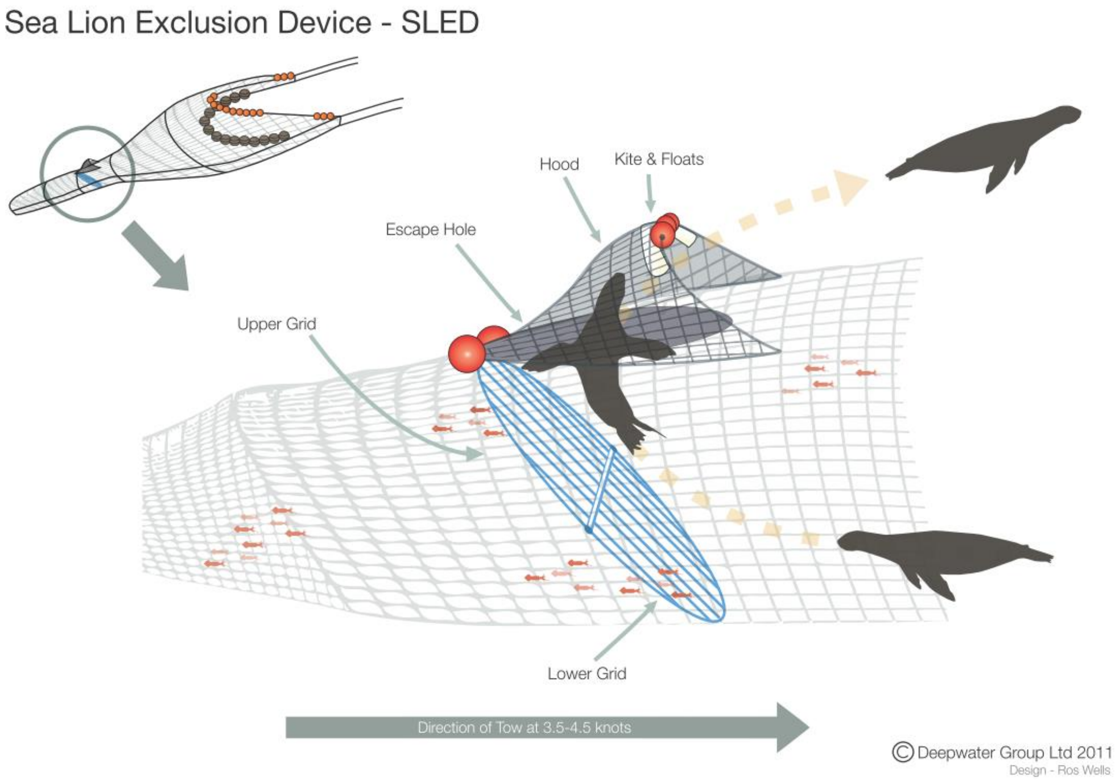

Figure 4.12: Diagram of a New Zealand sea lion exclusion device (SLED) inside a trawl net. Image courtesy of the Deepwater Group.

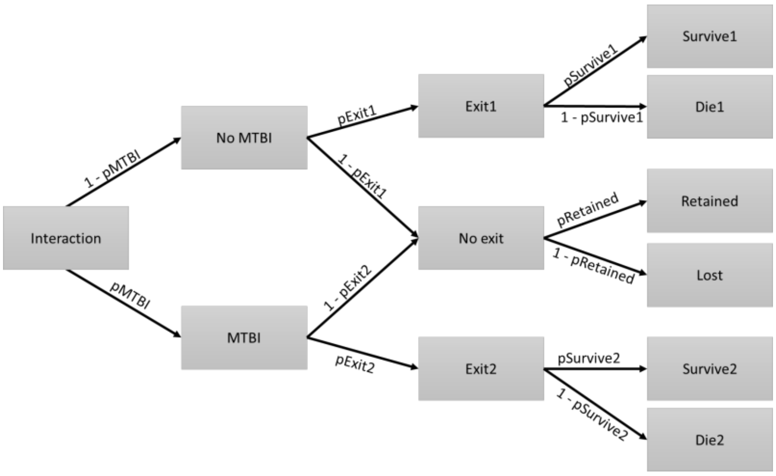

Figure 4.13: State transition process for New Zealand sea lions interacting with trawl nets that have SLEDs deployed, as developed under consultation with AEWG (November 2018). Boxes are categorical states; variables atop of arrows denote transition probabilities. MTBI is ‘mild traumatic brain injury’. See section 4.4.4.4 for details.

#### 4.4.4.1 MILD TRAUMATIC BRAIN INJURY

Sea lions may strike their heads on the SLED grid. To look for evidence of injury arising from such collisions, necropsies were conducted on animals recovered from cover net trials (see below) and on those incidentally caught and recovered from vessels operating in the SQU 6T, SQU 1T, and SBW 6I fisheries. All of several hundred New Zealand sea lions returned for necropsy died as a result of drowning rather than physical trauma from interactions with the trawl gear including the SLED grid (Roe & Meynier 2010, Roe 2010). Necropsies were designed to assess the nature and severity of trauma sustained during capture and to infer the survival prognosis had those animals been able to exit the net (Mattlin 2004). However, problems associated with this approach limited the usefulness of the results. For example, sea lions had to be frozen on vessels and stored for periods of up to several months before being thawed for 3–5 days to allow necropsy. Roe & Meynier (2010) concluded that this freeze-thaw process created artefactual lesions that mimic trauma but, particularly in the case of brain trauma, could also obscure real lesions. Further, two reviews in 2011 concluded that the lesions in retained animals may not be representative of the injuries sustained by animals that exit a trawl via a SLED (Roe & Meynier 2010, Roe 2010).

Notwithstanding the limitations of the necropsy data in assessing trauma for previously frozen animals, it was possible to determine that none of the necropsied animals sustained sufficient injuries to the body (excluding the head) to compromise survival (Roe & Meynier 2010, Roe 2010). However the potential for head trauma arising from impacts with the SLED grid could not be ruled out as a potential contributing factor to the animals' death (Roe & Meynier 2010, Roe 2010).

Abraham (2011) used biomechanical modelling, euphemistically referred to as 'crash-test-dummy' modelling, to quantify the likelihood of a sea lion experiencing physical trauma sufficient to render the animal insensible (and therefore likely to drown) arising from a collision with a SLED grid. This work used video footage of Australian fur seals interacting with comparable trawl exclusion devices (Lyle 2011, Wilcox 2008) to estimate (for sea lions) the likelihood of a head-first impact, the speed of impact, the angle of impact relative to individual SLED grid bars and relative to the grid plane, and the location of impact on the grid. The risk of MTBI was then assessed by biomechanical testing and modelling across a range of plausible and 'worst-case' impact scenarios (Ponte et al. 2010, 2011) and combined in a simulation-based probabilistic model (Abraham 2011). In the base case model, 2.7% of sea lions entering the trawl were estimated to experience MTBI; in the most extreme sensitivity the estimate was as high as 8.2%. These results indicate that rates of death by MTBI for New Zealand sea lions interacting with the SLED grid are likely to be low. It is thought that animals affected by MTBI may be more likely than uninjured animals to remain and drown in the net, where they will be counted among observed captures unless the body is subsequently lost from the net. For this reason MTBI may influence the rate at which sea lions exit or drown, but MTBI is not in itself a source of cryptic mortality.

#### 4.4.4.2 BODY NON-RETENTION

From first principles and considering SLED design (Figure 4.12) it seems unlikely that body non-retention rates are high, because:

i) the escape opening of SLEDs employed in New Zealand fisheries is at the top of the net, whereas drowned pinnipeds are observed to be negatively buoyant;
ii) forward-facing hoods are designed to allow exit for actively swimming animals and retain passive or inert bodies due to the forward motion of the net; and
iii) hood floats are designed to close the escape opening in the event that the trawl net becomes inverted (turns upside down) or when the net reaches the surface of the water.

Preliminary results of SLED monitoring trials in overseas jurisdictions support the conclusion that drowned pinnipeds are unlikely to be lost, and thereby not counted among observed captures, in trawls employing SLEDs. Overseas researchers with first-hand knowledge of the operation of these devices were consulted in the process of parameterising the cryptic mortality model (i.e., representing assumptions about body non-retention as a model input prior). Informed by expert input the Aquatic Environment Working Group estimated that between 1 and 10% of drowned sea lion carcasses may be subsequently lost from the net (i.e., a uniform prior on pRetention of 0.90 – 0.99).

#### 4.4.4.3 POST-ESCAPE DROWNING

Between 1999–2000 and 2002–03, an experimental approach was taken to estimate interaction rates and SLED efficacy rates, by intentionally capturing animals as they exited the escape hole of a SLED. Cover nets were added over the escape holes of some SLEDs and sea lions were restrained in these nets after they exited the SLED. An underwater video camera was deployed in 2001 to assess the behaviour and the likelihood of post-exit survival of those animals that were retained in the cover nets (Wilkinson et al. 2003, Mattlin 2004). Due to low sample sizes and ambiguous interpretation of necropsy results, this work was judged to be inconclusive (Roe 2010). Re-analysis of the video data in 2019 indicated that at least some of the animals were conscious and active at the time that they exited the net, but the number of observations was too low to draw any quantitative conclusions.

In 2019, data on sea lion dive behavior and trawl characteristics in the SQU 6T fishery were used to simulate the outcome of dives in which sea lions interact with SLED-equipped trawls, to estimate the probability of post-escape drowning (Middleton 2019b). This study used electronic telemetry data indicative of sea lion dive behavior under ‘normal’ foraging conditions to characterise critical rates such as: i) how long a sea lion can remain conscious underwater (euphemistically labeled ‘maximum breath-hold’ although pinnipeds actually expel their air before diving; oxygen is stored in the blood); ii) descent speed; iii) horizontal swimming speed; and iv) ascent speed. These data were available from sea lions tagged specifically at the Auckland Islands (Crocker et al. 2001, Chilvers et al. 2006).

The simulation tracked the fate of sea lions as they passed through the net using a time step whereby every minute underwater was subtracted from the animals’ remaining ‘breath-hold’ time (i.e., time at which the animal will run out of oxygen and become unconscious). The study used video data of Australian fur seals in nets equipped with comparable exclusion devices, to estimate the likelihood in each minute that an animal inside the net will exit via the SLED (Lyle & Wilcox 2008). Animals that contribute to cryptic mortality are those that fail to exit before becoming unconscious drown and are retained in the net; exit the net with sufficient time to reach the surface survive; exit the net, but with too little conscious time remaining to reach the surface, and are presumed to drown outside the net.

In the simulation base case, roughly 7% of animals exiting the net are estimated to nonetheless drown, and the ratio of total deaths to deaths that occur inside the net was estimated at 1.4 (range 1.2–1.5 depending on what proportion of the animals successfully exit, which varied between bottom trawl and midwater trawl nets.) These outputs were used to inform the definition of priors for the Bayesian estimation of cryptic mortality from all sources (Meyer 2019).

#### 4.4.4.4 BAYESIAN ESTIMATION OF SEA LION CRYPTIC MORTALITY IN TRAWLS USING SLEDS

New research was completed in 2019 integrating all available information indicative of the fate of sea lions that enter trawls equipped with SLEDs, in a Bayesian modelling framework (Meyer 2019). Models constructed under this project used a state transition matrix of different possible states for each sea lion that enters the net. The probability of each state transition was estimated by fitting to observed captures data, or was influenced by the priors (which were estimated outside the model or informed by expert knowledge).

The state transition matrix used to estimate cryptic mortality in sea lions is reproduced in Figure 4.13. Transition probabilities were informed as follows:

- the number of sea lions entering the net (annual interactions) were estimated outside the model (separately for midwater and bottom trawl fishery groups) as a function of spatial overlap between fishing effort and sea lion distribution (Large et al. 2019);
- the probability of a sea lion suffering MTBI was informed by ‘crash-test-dummy’ modelling described above, mean pMTBI = 0.027 (from Abraham 2011);
- the probability that a sea lion exits the net was fitted to observed captures — mean pExit = 0.57 for bottom trawls and 0.88 for midwater trawls;
- the probability that an exiting sea lion survives (i.e., does not drown before reaching the surface) was informed by simulation as described above (Middleton 2019b) — mean pSurvive = 0.94 for both bottom trawl and midwater trawl;

- the probability that the body of a sea lion that drowns in the net is retained was estimated by the AEWG, informed by expert discussion and observations of comparable exclusion devices for Australian fur seals (in which all drowned carcasses were retained in the net). Mean pRetention = 0.945 for both bottom trawl and midwater trawl.

Combining these estimates, the model estimates that the cryptic mortality multiplier is 1.15 in bottom trawls (95% c.i. 1.05–1.31) and 1.60 for midwater trawls (95% c.i. 1.20–2.63). These estimates were used to convert estimated captures to deaths in the risk assessment by Large et al. (2019).

Note that the apparent higher catchability in bottom trawls is offset by a lower cryptic mortality, suggesting sea lions are less likely to enter a bottom trawl relative to a midwater trawl, but also less likely to exit successfully via the SLED. Because these two factors work in opposition, the actual risk to sea lions per trawl event is similar between the two gear types.

Sensitivity analyses conducted by Meyer (2019) indicate that the model estimates of cryptic mortality are not highly sensitive to expert-derived assumptions (reflected as priors for the transition probabilities) within plausible ranges. The parameter with the most ability to affect the cryptic mortality multiplier is the probability that drowned carcasses are retained in the net, but major changes are required to this prior (e.g., assuming more than half of all drowned carcasses are lost from the net) to have large effects on the cryptic multiplier. High non-retention rates are not consistent with known hydrodynamic principles or observations of fur seal carcasses retained by comparable exclusion devices in Australia.

### 4.4.5 MODELLING POPULATION-LEVEL IMPACTS OF FISHERIES DEATHS

Consistent with terminology used in the SEFRA methodology (Chapter 3), Fisheries New Zealand has now adopted the term ‘Population Sustainability Threshold’ or PST to denote the number of anthropogenic deaths that a population can sustain while still meeting a defined population recovery or stabilisation outcome, evaluated via simulations using a demographic population model. The choice of reference outcome is a policy decision.

For Auckland Islands sea lions, the likely effect of fisheries impacts at different levels was estimated from a demographic population model informed by mark-recapture observations, annual pup census results, estimated fisheries-related deaths, and the estimated age distribution of lactating females, as described by Roberts & Doonan (2016). The model also supported a quantitative risk assessment to estimate the effects of non-fishery threats (section 4.4.5.1 below). In 2017, additional model runs were carried out under project SEA2026-30, incorporating the newest pup count data from Figure 4.3 (but not including all available mark-recapture data), and were used to update management options for the Operational Plan for the SQU 6T fishery in 2017 (Roberts 2017b).

The population projections by Roberts & Doonan (2016) relied on the untestable structural assumption that future demographic rates would approximate historically observed rates from the preceding 20 years. This period included the period of steepest population decline and subsequent apparent stabilisation, but not the preceding period of population growth. It is likely that changes in demographic rates reflect changing environmental conditions. Because it is not possible to anticipate what environmental conditions are likely to prevail in the future, with unknown potential consequences for sea lion demographic rates, uncertainty of this nature is best addressed with model sensitivities.

In 2019 the Auckland Islands sea lion demographic model of Roberts (2017b) was updated with all available data including mark-recapture information up to the 2018–19 field season (Roberts 2019). To address environmental uncertainty, the updated model was structured to estimate demographic rates separately based on periods of historically increasing (1990 to 1998), decreasing (1999 to 2009), and stable pup production trajectories (2010 to 2019) and to simulate the effects of fisheries mortality on population outcomes under these three regimes, reflecting that it is likely that decadal-scale climatic variability affects critical sea lion demographic rates, but future climate conditions cannot be foreseen.

Models also incorporated estimates of SQU 6T fishery related deaths, derived from the outputs of separate research projects estimating historical fishery captures (Large et al. 2019) and cryptic mortality levels relating to the use of sea lion exclusion devices (Meyer 2019).

The base case model produced good fits to pup census, mark-resighting, and age distribution observations. Model estimates indicate that the observed change in pup production trajectory in 2009 (from decreasing to stable) was driven by increased annual survival at age groups 2–5 y and 6–14 y, and a slightly increased annual pupping rate, rather than by improved first-year 'pup' survival, which was unchanged relative to the period of decline.

The base model estimated a current population size of 4293 females in 2019 (95% CI = 4120–4473), which was subsequently used in the estimation of the PST.

A new PST criterion was defined by Fisheries New Zealand, SEFRA approach (Chapter 3). A calibration coefficient ($\phi$) of 0.1 was selected as the base case value by Fisheries New Zealand, such that annual impacts equal to the PST are consistent with a stable population size at 95% of the un-impacted level. At this level of $\phi$, a female-only PST of 26 individuals was estimated. In comparison, Large et al. (2019) estimated 2.8 (median) or 5.4 (upper 95% CI) actual female deaths in the squid fishery from 2013 to 2017.

Assuming future fisheries mortality equal to the PST (at $\phi = 0.1$), i.e., 26 female deaths per year, model projections estimated a mature female population size in 2025 of between 95.0% (95% CI = 94.7–95.2%) and 96.1% (95% CI = 95.8–96.3%) of what would have occurred in the absence of fishery mortality (depending on the future population growth scenario). Under the 'stable' (recent) demographic rate scenario, future deaths consistent with recent estimated levels would result in a population size in 2025 of between 99.0% (95% CI = 99.0–99.1%) and 99.5% (95% CI = 99.5–99.5%) of un-impacted levels, depending on whether the upper 95% CI or median of recent annual deaths was assumed, respectively.

Note that because this modelling framework does not account for density dependence, these estimates of future population status will be lower than would be estimated from a generic application of the PST formulation using *Rmax* (equation 30 in Chapter 3). In that formulation, applying a value of ($\phi = 0.1$) yields an impacted population that is 5% lower than the un-impacted population in the long term (at equilibrium) because population productivity increases to compensate as the population is reduced below carrying capacity.

#### 4.4.5.1 MULTI-THREAT QUANTITATIVE RISK ASSESSMENT FOR AUCKLAND ISLANDS SEA LIONS

In 2016 a quantitative risk assessment estimating the potential impacts of both fisheries and non-fishery threats to sea lions was undertaken to inform the development of a Threat Management Plan for the species (Roberts & Doonan 2016). On the advice of the AEWG, for purposes of informing management, this model replaced the previous 'BFG model' that had been used and updated since 2000 (Breen et al. 2003, Breen & Kim 2006a, 2006b).

A panel of national and international independent experts, supported by relevant subject matter advisors, was convened to provide guidance on the level of threats to New Zealand sea lions and review the demographic assessment. The first of two workshops was held from 28 April to 1 May 2015. It built on previous discussions at a pup mortality workshop held in 2014, but considered all threats to all sea lion age groups. The initial stage of the risk assessment model – the demographic assessment – was completed in advance of the first workshop, for the panel to review and provide recommendations for model improvements (Debski & Walker 2016).

Separate demographic assessment models were developed for females at the Auckland Islands and Otago Peninsula, integrating information from mark-recapture observations, pup census, and the estimated age distribution of lactating females (Auckland Islands only). With respect to the Auckland Islands assessment, good fits were obtained to all three types of observation and the model structure and parameter estimates appeared to be a good representation of demographic processes that have affected population decline there (primarily low pup survival and low adult survival). The Otago Peninsula assessment made use of a much smaller number of observations, however still produced good estimates of all key demographic rates, with much higher pup survival relative to the Auckland Islands population (Roberts & Doonan 2016).

A two-stage assessment of the effects of threats was undertaken where the consequences of removing the effects of a threat was estimated in terms of the population growth rate of mature individuals in 2037. This used threat-specific mortality estimates at age (provided by MPI/DOC subsequent to two dedicated TMP workshops, see Debski & Walker 2016), in which:

1. 'triage' projections were undertaken for all assessed threats using the upper bound estimates of threat-related mortality to screen out threats that had little effect on projected growth rate;

2. 'best-estimate' projections were undertaken using the best estimate of threat-specific mortality for all threats that passed through the triage stage (Roberts & Doonan 2016).

The triage of the risks posed to New Zealand sea lions was conducted to limit the number of risks to be included in the more detailed Markov chain Monte Carlo (MCMC) modelling. To do this, a simple model was used to assess the upper bound, or worst case scenario, of the threat by predicting the response of the population to that threat being removed. The results of this triage are not considered to be the best estimate of the risks posed to the New Zealand sea lions, but a mechanism to reduce the list of the threats to those that have the largest influence.

Triage model run projection outputs for the Auckland Islands using the final model are shown in Figure 4.14 and for the Otago Peninsula population in Figure 4.15. The black line in Figures 4.14 and 4.15 indicates the estimated historical trend and population projection based on demographic parameters from the last 10 years. The removal of each single threat is plotted separately.

The effects of removing the threats that act on pups (i.e., *K. pneumoniae*, hookworm, wallows[5]) have a delayed effect on the size of the mature population of sea lions. This is because the pups that will survive still need time to mature before they are included in the modelled mature female population (Roberts & Doonan 2016, Debski & Walker 2016).

Removal of the upper bound of *Klebsiella* risk creates the largest change in population size over the 20-year time period (2017–37), however the population reacts more quickly to the removal of the upper bound of estimated trawl interactions because this acts directly on the mature females. The ratio of mature female population in 2037 compared with 2017 is 1.30 when *Klebsiella* is removed, and 1.24 when trawl interactions are removed (Roberts & Doonan 2016). The independent panel considered that some of the upper bounds used in the triage process were unlikely to be realistic and should be treated with caution (Debski & Walker 2016).

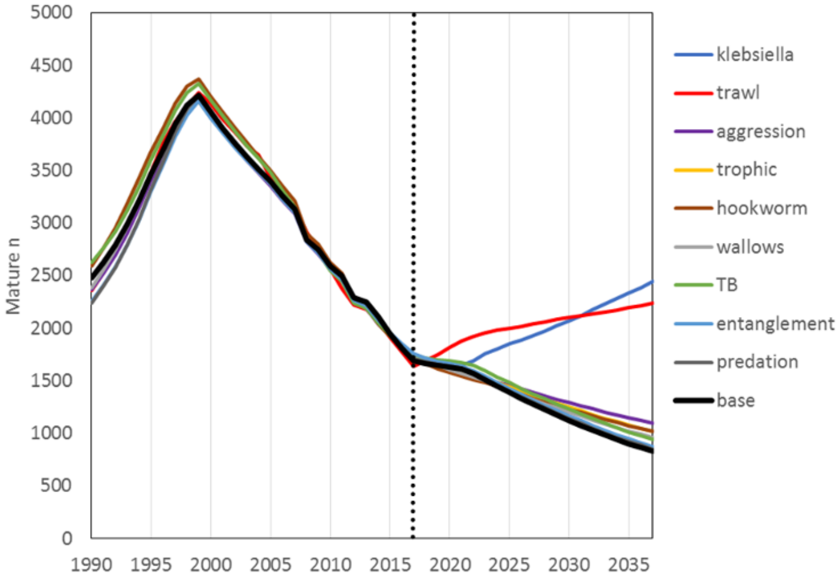

Figure 4.14: Triage projections of model estimated number of mature individuals (mature n) at the Auckland Islands during 1990–2037, using upper values of threat mortality. The black dotted line indicates the estimated historical trend and population projection based on demographic parameters from the last 10 years. The removal of each single threat is plotted separately as coloured lines. Reproduced from Roberts & Doonan (2016).

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[5] Where this report refers to this threat as 'wallows', this includes all types of hole, drop, or barrier that either cause a sea lion pup to be separated from its mother or to drown.

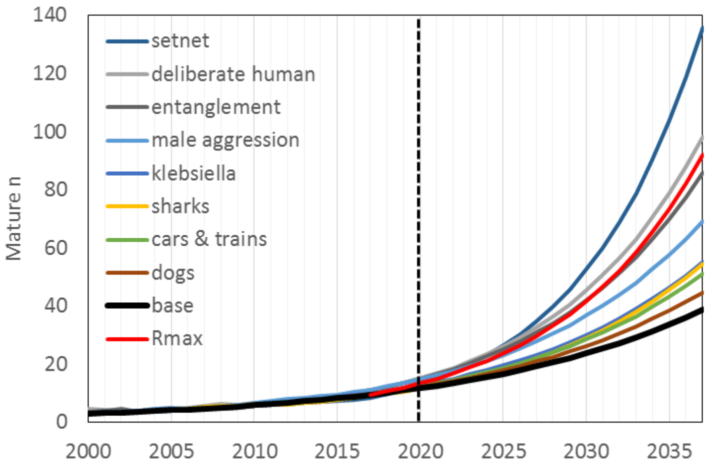

Figure 4.15: Triage projections of model estimated number of mature individuals (mature n) at the Otago Peninsula during 1990–2037, using upper values of threat mortality. The black dashed line indicates the estimated historical trend and population projection based on demographic parameters from the last 10 years. The removal of each single threat is plotted separately as coloured lines except for the red line, which shows population growth at $R_{\max}$ (assumed to be 0.12). Reproduced from Roberts & Doonan (2016).

For the Otago Peninsula model, the removal of upper bounds of some threats produced a very rapidly growing population, higher than the assumed maximum optimal growth rate ($R_{\max}$) (Figure 4.15). This indicates that the upper bounds used for set net and deliberate human threats were probably unrealistically high (Roberts & Doonan 2016, Debski & Walker 2016).

For the Auckland Islands population, best-estimate projections were undertaken for commercial trawl related mortality, *Klebsiella* related mortality of pups, trophic effects (food limitation), pups drowning in wallows, male aggression, and hookworm mortality. These threats were compared with the base run — a continuation of demographic rates since 2005 ($\lambda_{2037}$ = 0.961, 95% c.i.: 0.890–1.020). A positive growth rate was obtained only with the alleviation of *Klebsiella* ($\lambda_{2037}$ = 1.005, 95% c.i.: 0.926–1.069). When assuming the most pessimistic view of cryptic mortality (all interactions resulted in mortality and associated death of pups), alleviating the effects of commercial trawl related mortality resulted in an increased population growth rate relative to the base run, but did not reverse the declining trend ($\lambda_{2037}$ = 0.977, 95% c.i.: 0.902–1.036) (Figure 4.16). The alleviation of trophic effects (food limitation) had the next greatest effect ($\lambda_{2037}$ = 0.974, 95% c.i.: 0.905–1.038), and all other threats had a minor effect relative to the base run projection (increase in $\lambda_{2037}$ of less than 0.01) (Figure 4.17, Roberts & Doonan 2016).

For the Otago Peninsula population, similar effects were estimated with the alleviation of any of the threats that passed through triage: commercial set net fishery related mortality, direct human mortality, pollution related mortality, entanglement, and male aggression, relative to the base run projection ($\lambda_{2037}$ = 1.070, 95% c.i.: 1.053–1.087). Deliberate human mortality was estimated to have the greatest effect on projected population size ($\lambda_{2037}$ = 1.093, 95% c.i.: 1.075–1.112) (Figure 4.18, Roberts & Doonan 2016).

For the Auckland Islands population, model outputs suggest that if demographic rates used to simulate forward population trajectories (i.e., sampled from the previous 20-year period) are accurate, then the TMP goals would be difficult to achieve with the complete alleviation of a single threat. In this context, the most effective approach to meeting the goals of the TMP may be to spread the management effort across the suite of key perceived threats identified from this assessment.

The population projections are sensitive to assumptions about what demographic rates are being realised in the population, in the context of considerable environmental variability on a decadal scale, with likely effects on critical demographic rates driving population change. A high priority is the development of tools for monitoring the effects of environmental and management drivers on threat-specific mortality and influential demographic rates

(Roberts & Doonan 2016). For example research to examine factors affecting pup survival (Edwards & Roberts in review; Roberts et al. in review) commenced in 2018, and integrative ecosystems research to investigate decadal scale climate variability potentially affecting sea lion demographic parameters is being progressed under a separate contract (ZBD2018-05).

The assessment for some of the key threats to New Zealand sea lions was hampered by incomplete information for estimating threat-specific mortality, e.g., relating to the causes of pup mortality during the entire first year of life and of cryptic mortality. In addition, a lack of demographic observations for the Campbell Island and Stewart Island populations precluded the development of comprehensive quantitative risk assessments for these populations (Roberts & Doonan 2016).

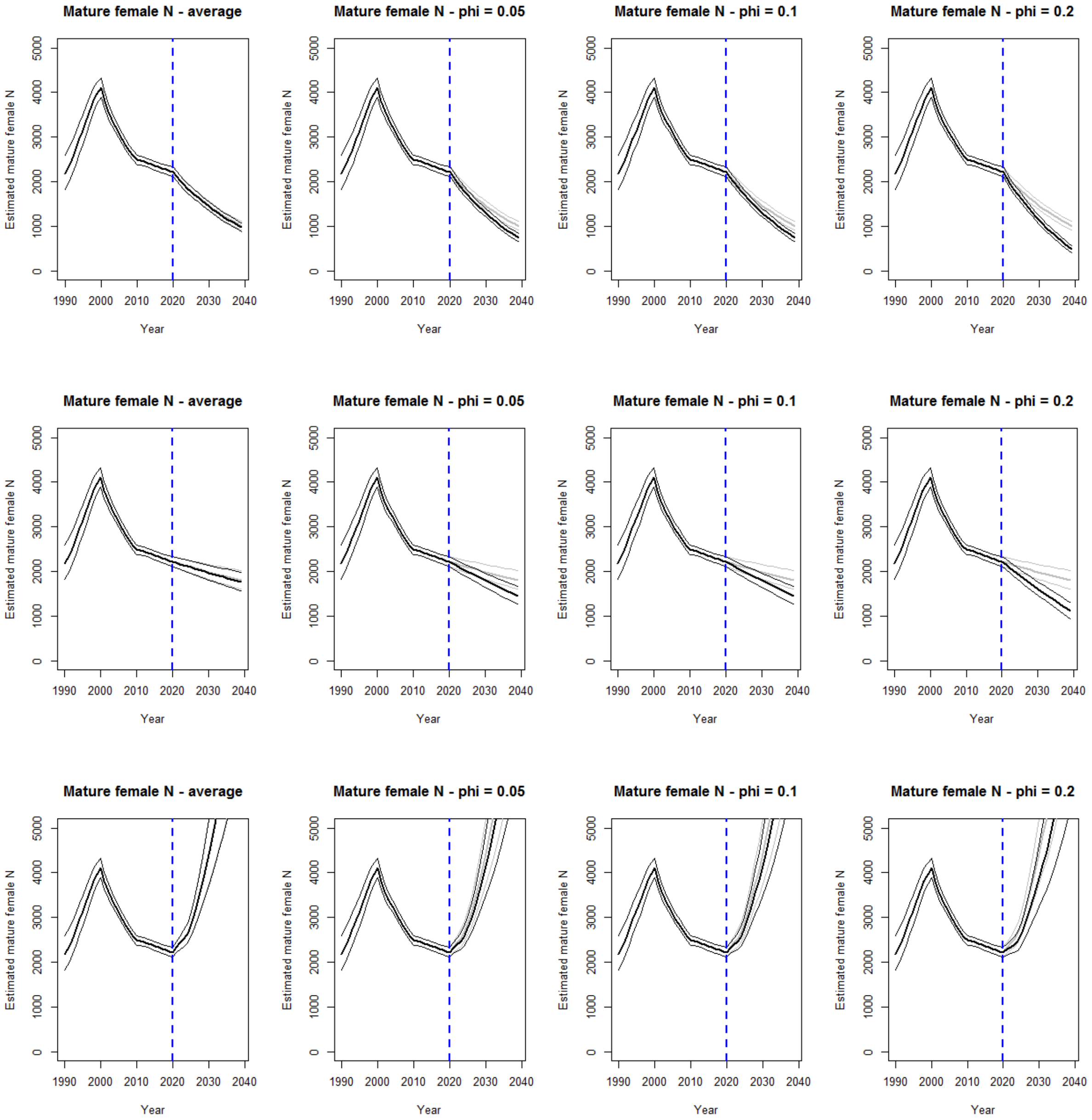

Figure 4.16: Female New Zealand sea lions population projections with alternative scenarios of: demographic rates affecting population growth (i.e., decreasing (top row), stable (middle row), or increasing (bottom row)); and future squid fishery-related deaths (i.e., zero future deaths (grey, shown in all plots for reference), the average of the last five years of estimated deaths, or equal to the population sustainability threshold (PST) assuming alternative values of $\phi$ (“phi”, p. 25 equation 30) (all black)). Reproduced from Roberts (2019).

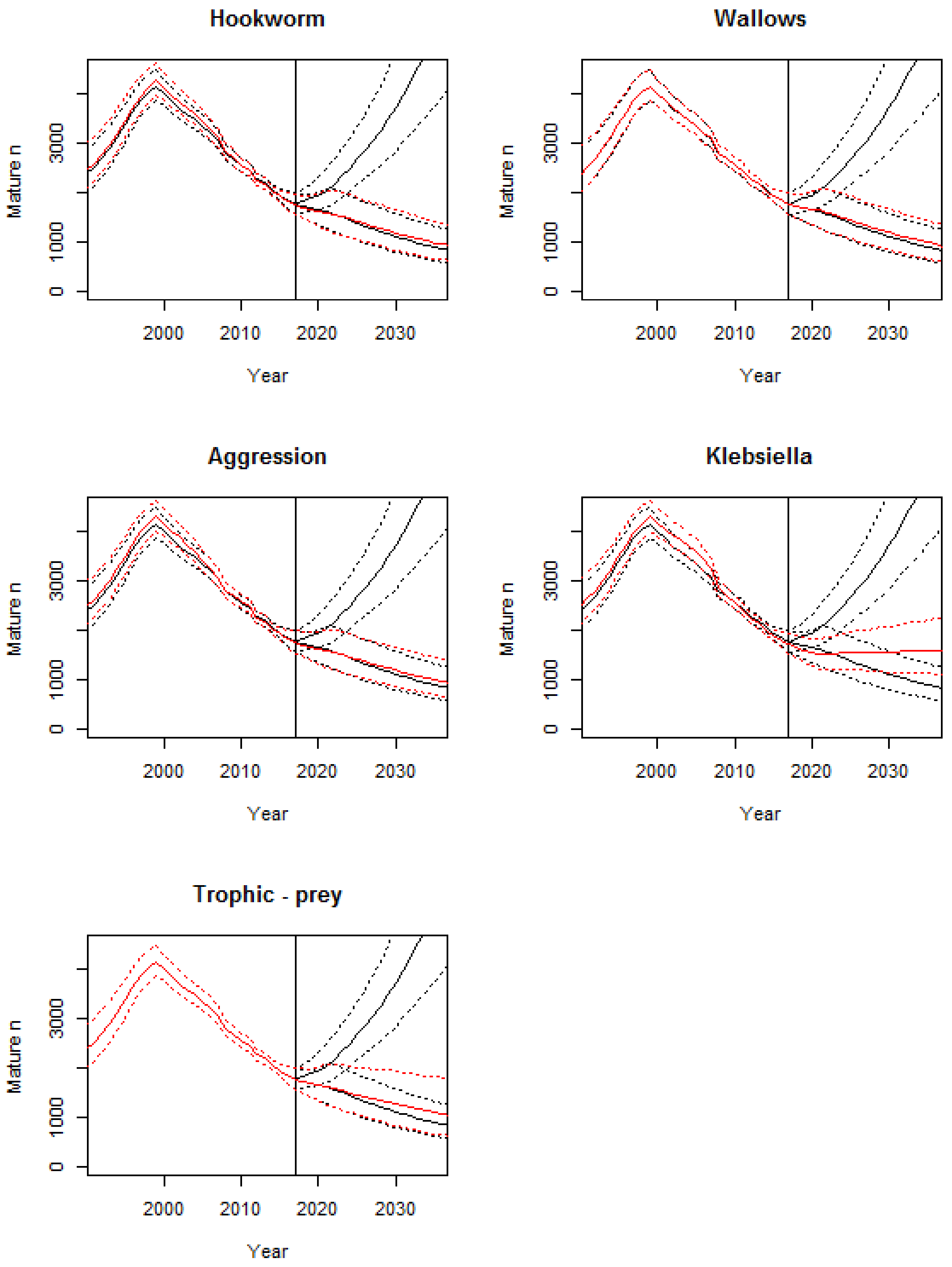

Figure 4.17: Best-estimate projections of mature individuals (mature n) at the Auckland Islands in the period 1990–2037 for all other threat scenarios. Lower black lines are with all threats (base run); upper black lines are with the 'max growth' scenario (1990–93 estimate of Surv0, 1990–98 estimates of Survival 6–14 years of age, and 1990–99 estimate of PrP; red lines are with a threat alleviated. Reproduced from Roberts & Doonan (2016).

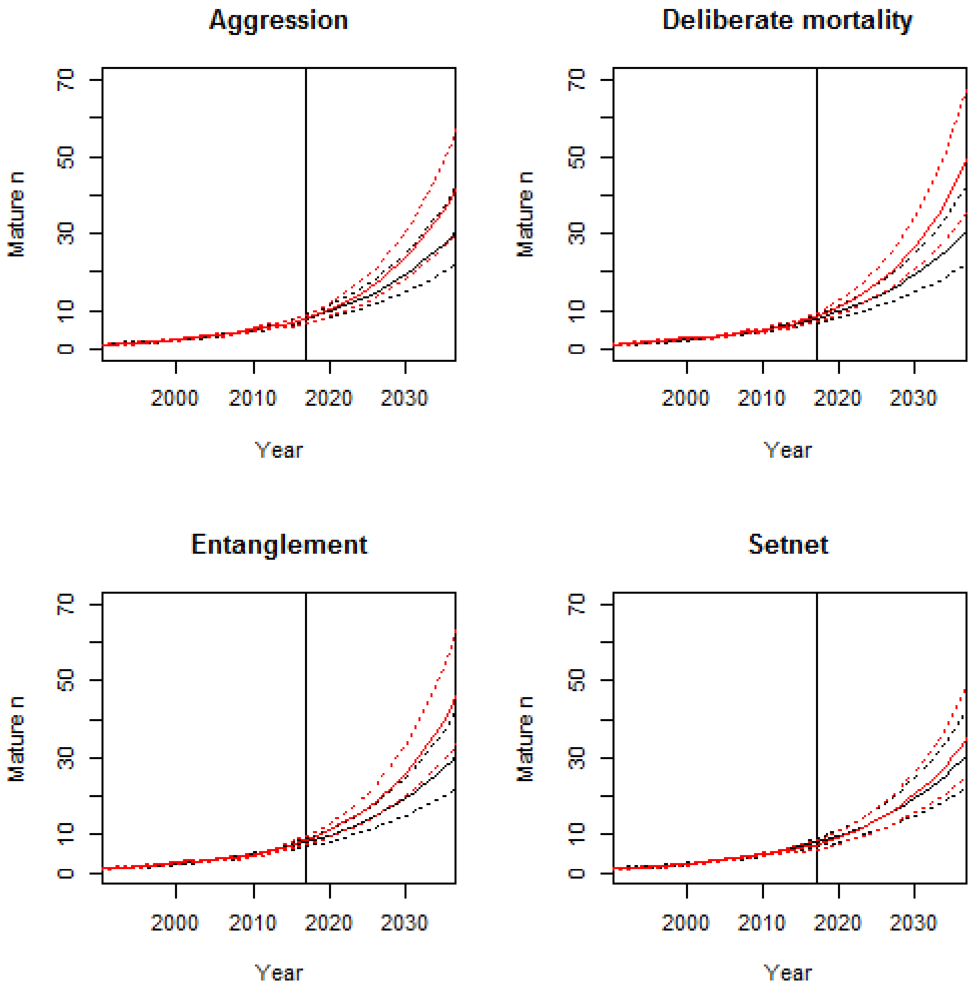

Figure 4.18: Best-estimate projections of mature individuals (mature n) at the Otago Peninsula in the period 1990–2037 for all threat scenarios (from Roberts & Doonan 2016). Black lines are with all threats (base run); red lines are with the threat alleviated. Reproduced from Roberts & Doonan (2016).

### 4.4.5.2 PBR (NOW PST) ASSESSMENT FOR CAMPBELL ISLAND POPULATION

Following the 2013 season in which a hjgh number of New Zealand sea lions were captured in the Campbell Rise Southern blue whiting fishery (SBW 6I), a review was conducted of potential biological removal (PBR) guidelines and relevant scientific literature to inform the selection of appropriate PBR parameter values for the Campbell Island sub-population (Roberts et al. 2014b). The PBR is a traditional approach to defining a safe level of human-related mortalities of marine mammals, which was originally developed for the US Marine Mammals Protection Act (Wade 1998). It is a precursor to the PST formulation used for most New Zealand protected species; like the PST, PBR relies on $R_{max}$ to represent the species intrinsic population growth rate, but rather than full Bayesian consideration of parameter and modelling uncertainty it uses a conservative point estimate of population size $N_{min}$ and a recovery factor $F_R$ that provides for parameter bias correction as well as reflecting policy decisions regarding the level of acceptable impact (replacing the calibration factor phi in the PST formulation). New work is planned to replace this PBR estimation with an updated PST estimate consistent with the New Zealand SEFRA framework.

The pup census at Campbell Island of 681 pups in 2010 (Maloney et al. 2012) was taken as a robust lower estimate of total pup production. A matrix modelling analysis was conducted to estimate plausible pup to whole-of-population multipliers of 4.5 and 5.5, which were applied to the pup census estimate to calculate $N_{min}$ values of 3065 and 3746. The rate of increase in pup counts from a time series of pup censuses was used as an approximation to whole-of-population growth rate for estimating a credible lower limit of $R_{max}$. Values of 0.06, 0.08, and 0.10 were used in PBR calculations, with the upper and lower limits considered as plausible bounds for this parameter used in a sensitivity analysis. The Auckland Islands and Campbell Island sea lion breeding populations are likely to be demographically independent, so were assessed as separate subpopulations (Moore & Merrick 2011). A default recovery factor ($F_R$) of 0.5 was applied, as is used in the USA for stocks of a threatened species with unknown (or not declining) population trajectory. (Roberts et al. 2014b).

Prior to 2005–06 the annual number of captures was very low, though capture rate appears to have increased since, with the greatest number of captures in 2012–13 (Table 4.7). Running means of capture levels (3 and 5-year) were also calculated for comparison with PBR estimates. For $F_R$ of 0.5, and the selected estimates of $N_{min}$ (3065) and $R_{max}$ (0.08), the calculated PBR was 61. Estimated captures did not exceed the PBR in any year when the default $F_R$ of 0.5 was used, regardless of which other parameter values used. When the lower $F_R$ of 0.1 was used, the calculated PBR of 12 was exceeded in two years when using a 3-year running mean of captures and in one year with a 5-year running mean of captures. When $F_R$ of 0.2 was used, the calculated PBR of 25 was not exceeded in any year. There has been a very strong bias towards males in observed captures (Thompson et al. 2013). An array of female-only PBRs was estimated by halving the PBR for all animals and was not exceeded by female captures in any year regardless of which combination of parameter values was used (Roberts et al. 2014b).

New work is planned by Fisheries New Zealand under project PMM2019-09 to re-estimate a Campbell Island PST incorporating updated demographic information.

### 4.4.6 MANAGING FISHERIES RISK

For New Zealand sea lions, efforts to mitigate incidental captures in fisheries have historically focused on the SQU 6T fishery.

Current management reflects previously designated spatial fisheries closures. In 1982 the Minister of Fisheries established a 12-nautical mile exclusion zone around the Auckland Islands from which all fishing activities are excluded (Wilkinson et al. 2003); in 1995, the exclusion zone was replaced with a Marine Mammal Sanctuary with the same controls on fishing (Chilvers 2008). The area was subsequently designated as a Marine Reserve in 2003.

From 1992, the Ministry adopted a maximum allowable level of fisheries-related mortality (MALFiRM; later referred to as a Fisheries Related Mortality Limit, FRML) to set an upper limit on the number of New Zealand sea lions that can be incidentally killed each year in the SQU 6T trawl fishery (Chilvers 2008). If this limit is reached, the fishery is closed for the remainder of the season. The original MALFiRM was calculated using the 'potential biological removals' approach (PBR; Wade 1998) and was used from 1992–93 to 2003–04 (Smith & Baird 2007a). From 2003–04 to 2017–18 the FRML was translated into a fishing effort limit (maximum permitted number of tows) based on assumptions about the interaction rate and SLED efficacy rate, regardless of the number of observed New Zealand sea lion captures. This approach was taken because since the introduction of SLEDs, observed sea lion captures were no longer a reliable index of the number of sea lions interacting with the net, and there was uncertainty about the survival rate of sea lions exiting the net via the SLED.

SLEDs were first deployed and trawled on some vessels in the SQU 6T fishing fleet from around 2000. SLED use increased in subsequent years through a development phase in which SLED designs were trialed and modified, followed by a phase in which a single design specification existed but was not mandated or universally adopted on all fishing effort across the fleet. Subsequently the Squid Fishery Management Company in consultation with Fisheries New Zealand mandated a standardised SLED design that would be required for the vessel to receive the 'discount rate' relative to the tow limit applied by the government (set to ensure that estimated mortalities remained below the designated FRML; see section 4.4.6.1, below). From the 2008 season the standardised model Mark 3/13 SLED (Figure 4.12) has been universally employed by all vessels in the SQU 6T fleet. SLED design consistent with these specifications, and SLED deployment during fishing operations, are audited and monitored by Fisheries New Zealand Observers.

From 2017, advice to manage sea lion interactions in this fishery was developed in consultation with the Squid 6T Operational Plan Technical Advisory Group (SqOPTAG), including representatives from government and stakeholder groups as well as technical experts and advisors. Under an Operational Plan adopted in December

2017, Fisheries New Zealand set an FRML for sea lions in the SQU 6T fishery based on estimation of a Population Sustainability Threshold (PST) using a Bayesian population dynamic model (Roberts & Doonan 2016). The PST represents the maximum number of anthropogenic mortalities that the population can sustain while still achieving a defined population objective. For the Auckland Islands sea lion population, the choice of population objective underlying the PST was as follows: 'Fisheries mortalities will be limited to ensure that the impacted population is no more than 5% lower than it would otherwise be in the absence of fishing mortality, with 90% confidence, over five years'. The choice of the population objective is a policy decision.

The SQU 6T Operational Plan was updated in 2019 reflecting the outcomes of the new scientific approach whereby interactions, captures, and deaths (including cryptic mortality) are estimated directly and observed captures are applied toward the adopted FRML without the need for a proxy effort limit. The outputs of the new scientific approach were reviewed and the implications for advice to inform an updated management plan were discussed via the SqOPTAG.

The four-year Squid 6T Operational Plan was adopted in 2019 and will remain in place until 30 September 2023[6]. The Operational Plan defines a new FRML to reflect updated population model outputs, including sensitivities reflecting the likelihood that critical demographic rates for Auckland Islands sea lions are affected by decadal scale climatic variations (Roberts 2019, above). The plan also sets a minimum observer coverage requirement of 90%, to ensure that sea lion captures are recorded and SLEDs are properly deployed.

#### 4.4.6.1 MANAGEMENT SETTINGS IN THE SQUID 6T FISHERY

Before the widespread use of SLEDs, New Zealand sea lions incidentally caught during fishing were usually retained in trawl nets and hauled onboard, allowing observers to gain an accurate assessment of the number of New Zealand sea lion interactions on observed tows in a given fishery. This enabled a robust estimation of the total number of New Zealand sea lions killed by fishing. However, following the introduction of SLEDs, the number of New Zealand sea lions interacting with trawls but exiting via the SLED was unobservable, so the interaction rate was instead estimated statistically. Subsequently, a management setting meant to approximate the interaction rate, i.e., the 'strike rate' was set by the government (along with a second setting, the 'discount rate' representing SLED efficacy, see below) to inform a proxy estimate of potential sea lion fatalities per 100 tows. This proxy estimate was then used to set an effort limit (maximum number of tows) on the operation of the fishery, to ensure that sea lion fisheries mortalities remained below the FRML.

The 'discount rate' was a management setting that approximated SLED efficacy, i.e., the proportion of sea lion interactions in which the sea lion exits the SLED and survives. The management regime for the SQU 6T fishery provided that the discount rate would be applied to all tows in which an approved Mark 3/13 SLED was used and relevant requirements of the Operational Plan met (e.g., notification of intention to fish in SQU 6T and reporting requirements). Discount rates applied between 2003–04 and 2018–19 are shown in Table 4.10.

The SLED discount rate was a fisheries management setting and should not be confused with the actual estimated survival rate of New Zealand sea lions exiting the SLED; for example the discount rate could be set deliberately lower than the actual estimated SLED efficacy rate, reflecting cautious management in the presence of uncertainty.

From 2019 a new science approach was adopted under which sea lion interactions, captures, and deaths (including cryptic mortality) are estimated directly. Under this approach it is now possible to evaluate performance against the FRML using observed captures directly, without the need for an effort proxy and associated SLED discount rate. For this reason, the new Squid 6T Operational Plan does not define a strike rate or discount rate; instead, total captures are monitored by fisheries observers and compared against the FRML as the season progresses. Cryptic deaths are estimated as a proportion of observable deaths, effectively adjusting the capture limit lower to account for sea lion bodies that may not be counted by fisheries observers.

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[6] https://www.fisheries.govt.nz/dmsdocument/38189-squid-6t-operational-plan-2019-2023

Table 4.10: Maximum allowable level of fisheries-related mortality (MALFiRM) or fisheries-related mortality limit (FRML) from 1991 to 2015. Note that direct comparisons among years of the limits in Table 4.10 are not possible because the assumptions underlying the MALFiRM or FRML changed over time.

| Year | MALFiRM or FRML | Discount rate | Management actions |
|---|---:|---:|---|
| 1991–92 | 16 (female only) | | |
| 1992–93 | 63 | | |
| 1993–94 | 63 | | |
| 1994–95 | 69 | | |
| 1995–96 | 73 | | Fishery closed by MFish (4 May) |
| 1996–97 | 79 | | Fishery closed by MFish (28 March) |
| 1997–98 | 63 | | Fishery closed by MFish (27 March) |
| 1998–99 | 64 | | |
| 1999–00 | 65 | | Fishery closed by MFish (8 March) |
| 2000–01 | 75 | | Voluntary withdrawal by industry |
| 2001–02 | 79 | | Fishery closed by MFish (13 April) |
| 2002–03 | 70 | | Fishery closed by MFish (29 March), overturned by High Court |
| 2003–04 | 62 (124) | 20% | Fishery closed by MFish (22 March), overturned by High Court FRML increased |
| 2004–05 | 115 | 20% | Voluntary withdrawal by industry on reaching the FRML |
| 2005–06 | 97 (150) | 20% | FRML increased in mid-March due to abundance of squid |
| 2006–07 | 93 | 20% | |
| 2007–08 | 81 | 35% | |
| 2008–09 | 113 (95) | 35% | Lower interim limit agreed due to the decrease in pup numbers |
| 2009–10 | 76 | 35% | |
| 2010–11 | 68 | 35% | |
| 2011–12 | 68 | 35% | |
| 2012–13 | 68 | 82% | |
| 2013–14 | 68 | 82% | |
| 2014–15 | 68 | 82% | |
| 2015–16 | 68 | 82% | |
| 2016–17 | 68 | 82% | |
| 2017–18 | 38 | 75% | |
| 2018–19 | 38 | 75% | |
| 2019–20 | 52 | N/A | New approach whereby deaths are estimated directly as a function of captures, eliminating the need for an effort limit and discount rate setting |

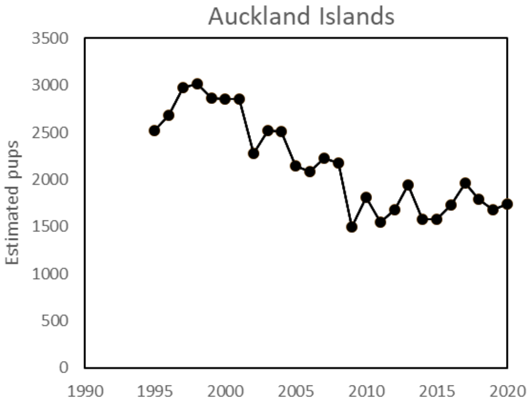
## 4.4.7 KEY INFORMATION GAPS

The Roberts & Doonan (2016) model and subsequent updates make no assumptions about the current status of the Auckland Islands sea lion population relative to ecological carrying capacity. Previously a review of life-history traits such as pup mass, pup survival, and female fecundity found no evidence for density dependent responses in the Auckland Islands population (Chilvers 2012b). However a number of indicators of nutritional stress have been identified during the period of population decline, including a temporal shift in diet composition to small-sized prey (Childerhouse et al. 2001, Stewart-Sinclair 2013), low pupping rate/delayed age at first pupping (Childerhouse et al. 2010a, Roberts et al. 2014a), low pup/yearling survival rate (Roberts et al. 2014a) and reduced maternal condition (Riet-Sapriza et al. 2012, Roberts & Doonan 2014) – all of which are common density dependent responses. However there is no evidence of typical density dependent responses, such as poor pup survival, being alleviated with decreasing population size (Roberts 2019). The underlying environmental causes of the apparent change is unknown; and it is unknown whether similar changes can be expected in future, and on what time scales. For this reason updates of the demographic model in Roberts (2019) estimated population trajectories under three different hypothetical climatic regimes corresponding to observed periods of growth, decline, and stability in the annual pup production trend, and additional work is underway under project ZBD2018-05 to better understand potential climatic drivers of ecosystem change including potential ‘regime shifts’ affecting sea lions. Analysis of factors affecting pup survival and subsequent effects on demographic rates (e.g., Edwards & Roberts in review; Roberts et al. in review) may yield additional insight.

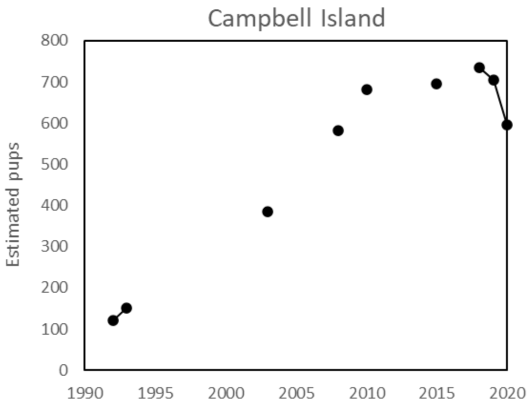
The spatial risk model of Large et al. (2019) relies on a single spatial foraging density layer informed by telemetry data collected primarily in summer, and primarily from breeding females at the Sandy Bay colony. The seasonal bias is appropriate for summer fisheries (i.e., targeting squid) and the sex and age bias is appropriate for a risk model concerned primarily with modelling the effect of fisheries on population reproductive output. However the seasonal bias toward summer may affect the accuracy of risk estimates for winter fisheries such as scampi, and the relative lack of data from the Dundas Island and Figure of Eight Island colonies may introduce other biases. New data collection is proposed to address these gaps.

## 4.5 INDICATORS AND TRENDS

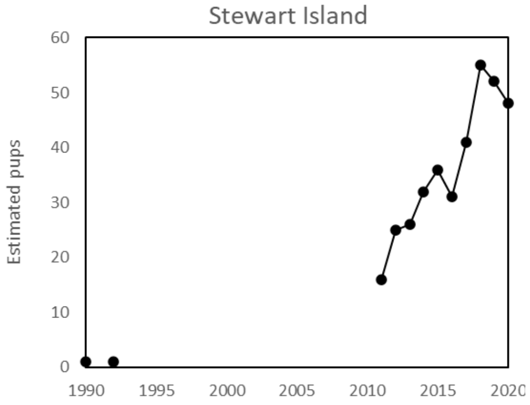
<table>
<tr>
<td><em>Population size</em></td>
<td>Roberts & Doonan (2016) estimated 11 755 New Zealand sea lions including pups (immediately after pupping) across all populations.<br>It is estimated that there were: 1740 pups born at the Auckland Islands in 2019–20 (Melidonis & Childerhouse 2020); 595 pups born at Campbell Island in 2019–20 (McNutt et al. 2020); 48 pups born at Stewart Island in 2019–20 (47 at Port Pegasus and 1 at Ulva Island; DOC unpublished data); and 21 pups born on the Otago coast in 2019–20 (DOC unpublished data).</td>
</tr>
<tr>
<td><em>Population trend</em></td>
<td>Estimated annual pup production at the Auckland Islands, Campbell Island, Stewart Island, and New Zealand South Island is shown below. Note that the y-axis scale varies in each plot.



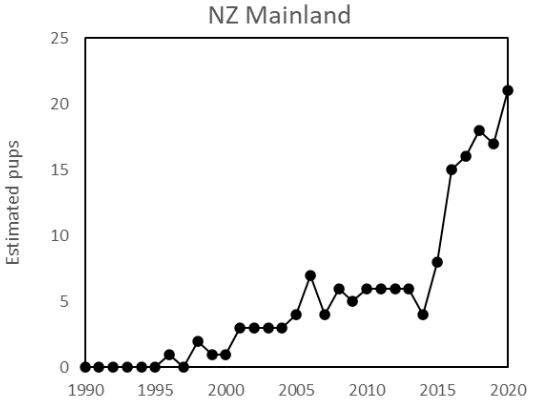
Auckland Islands



Campbell Island



Stewart Island



NZ Mainland

</td>
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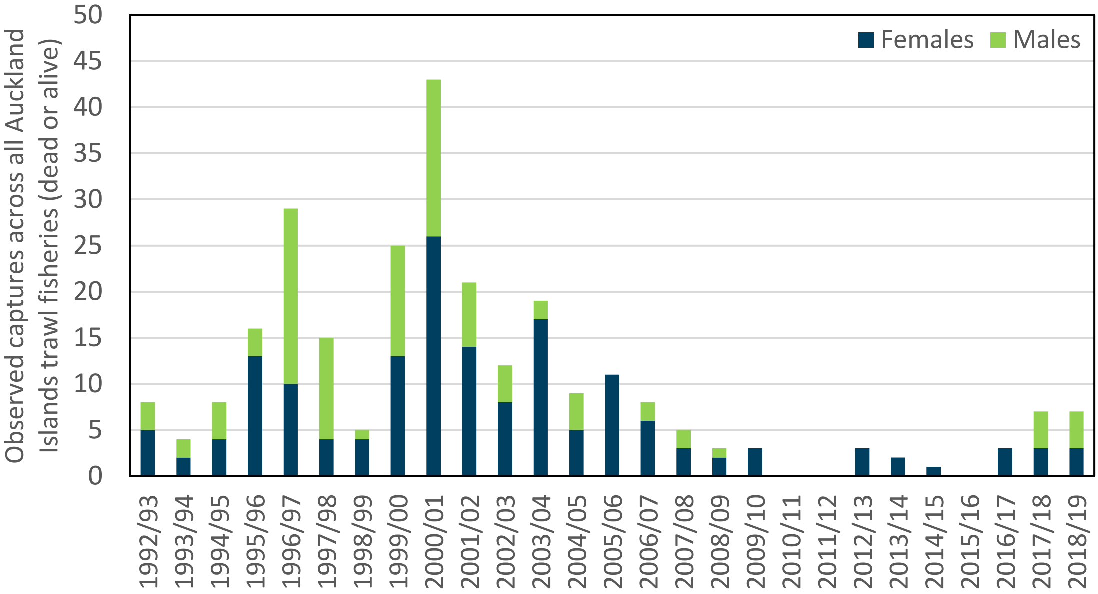
</table>

| | |
|---|---|
| *Threat status* | New Zealand: Nationally Vulnerable, Criterion D(1)[7], Range Restricted[8], in 2019[9]<br>IUCN: Endangered, A4bd[10], in 2015 |
| *Number of captures* | 3 observed captures in trawl fisheries in 2016–17<br>7 observed captures in trawl fisheries in 2017–18<br>7 observed captures in trawl fisheries in 2018–19 |
| *Trends in observed captures (both sexes)* |  |
| *Trends in estimated deaths (females only). From Large et al. (2019).* |  |

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[7] A taxon is listed as ‘Nationally Vulnerable’ under criterion D1 if it has a large population (5000–20 000 mature individuals) and a moderate to high rate of ongoing or predicted population decline (-30 to -70%) over three generations.

[8] A taxon is listed as ‘Range Restricted’ if it is confined to specific substrates, habitats, or geographic areas of less than 1000 km$^2$ (100 000 ha); this is assessed by taking into account the area of occupied habitat of all subpopulations (Townsend et al. 2008).

[9] Baker et al. (2019).

[10] A taxon is listed as ‘Vulnerable’ if it is considered to be facing a high risk of extinction in the wild. A3b refers to a reduction in population size (A), based on a reduction of 30% or more over the last 10 years or three generations (whichever is longer up to a maximum of 100 years (3); and when considering an index of abundance that is appropriate to the taxon (b; IUCN 2010).

## 4.6 REFERENCES

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