Assessment, evaluation and mitigation of marine mammal bycatch in commercial fishing gear

by

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Declaration of originality

This thesis contains no material which has been accepted for a degree or diploma by the University or any other institution, except by way of background information and duly acknowledged in the thesis, and to the best of my knowledge and belief contains no material previously published or written by another person except where due acknowledgement is made in the text of the thesis, nor does the thesis contain any material that infringes copyright.

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Candidate was the primary author. Author 1 contributed to the conception, design and development of the research project and edited drafts. Candidate contributed approximately 75% to the conception, design, planning, execution and preparation of the work for the paper.

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Associated publications

During my PhD candidature, I was also involved in the publication of the following papers or reports that are relevant to this thesis and/or to bycatch mitigation:

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Baker, G. B. and **Hamilton, S.** (2016). Risk of Unacceptable Impacts to the Populations of Incidentally Caught Seabirds from Small Vessel (<34 m) Ling Bottom Longline Fisheries (QMAs 3-7). Report prepared for Deepwater Group Ltd, New Zealand.

FAO. (2018). *Expert workshop on means and methods for reducing marine mammal mortality in fishing and aquaculture operations. Rome, 20-23 March 2018. Food and Agriculture Organisation of the United Nations (FAO)*. Retrieved from http://www.fao.org/3/I9993EN/i9993en.pdf (**S. Hamilton** one of 24 international, invited expert participants and contributors to this workshop and report).

Hamilton, S., Baker, G.B., Haward, M., Lea M-A. and Terauds, A. (2019). Review and analysis of underwater video of seal interactions with commercial trawl gear fitted with Seal Exclusion Device. University of Tasmania, Institute for Marine and Antarctic Studies, Hobart, Tasmania, Project 112378. Report prepared for the Australian Fisheries Management Authority (AFMA) Contract Reference ID PO1769.

Roberts, J., Childerhouse, S., Roe, W., Baker, G. B. and **Hamilton, S.** (2018). No evidence of cryptic bycatch causing New Zealand sea lion population decline. *Proceedings of the National Academy of Sciences, 115*, E8330–E8331. https://doi.org/10.1073/pnas.1806136115

Dedication

In loving memory of my dad, Ian, and my brother, Stu.

"… run like a river runs to the sea…"

U2 'One Tree Hill' 1987

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Contents

Declaration of originality i
Authority of access i
Statement of authority of access i
Statement of Co-Authorship ii
Associated publications iv
Dedication v
Acknowledgements vi
Thesis Abstract xiv

Chapter 1 — General Introduction 1

Chapter 2 — Technical mitigation to reduce marine mammal bycatch and entanglement in commercial fishing gear: lessons learnt and future directions... 7

2.1 Abstract 7
2.2 Introduction 8
2.3 Methods and scope 9
2.4 Results of reviewed technical mitigation measure 10
2.4.1 Mitigation relevant to multiple types of fishing gear 14
Pingers (Acoustic Deterrent Devices) 14
Weakened gear 17
2.4.2 Trawl 18
Net colour 18
Net binding 18
Exclusion devices with separation grids 19
Rope or mesh barriers 20
Auto-trawl systems 21
2.4.3 Purse seine 21
2.4.4 Longline 22
Passive acoustic deterrents 22
Catch protection devices – demersal longline 22
Triggered catch protection devices – pelagic longline 23
2.4.5 Gillnet 24
Acoustically reflective nets 24

Visually detectable nets 25
"Buoyless" nets 25
2.4.6 Pot/Trap 25
Pot/trap excluder devices 26
"Seal socks" 26
Sinking groundlines 26
Rope-less pot/trap systems 26
Rope colour changes 27
Stiff ropes 27
2.5 Conclusions and recommendations 27
Trawl – conclusions and research needs 27
Purse seine – conclusions and research needs 29
Longline – conclusions and research needs 29
Gillnet – conclusions and research needs 30
Pot/Trap – conclusions and research needs 31
Final summary and conclusions 32
2.6 Appendices 34
Appendix 2A: Technical mitigation measures developed to reduce marine mammal bycatch in trawl fishing operations. 34
Appendix 2B: Technical mitigation measures developed to reduce marine mammal bycatch in purse seine fishing operations. 42
Appendix 2C: Technical mitigation measures developed to reduce marine mammal bycatch in longline fishing operations. 44
Appendix 2D: Technical mitigation measures developed to reduce marine mammal bycatch in gillnet fishing operations. 48
Appendix 2E: Technical mitigation measures developed to reduce marine mammal bycatch in pot and trap fishing operations. 56
Chapter 3 — Review of research and assessments on the efficacy of sea lion exclusion devices in reducing the incidental mortality of New Zealand sea lions *Phocarctos hookeri* in the Auckland Islands squid trawl fishery 67
3.1 Abstract 67
3.2 Introduction 68
3.3 Methods 74
3.3.1 Review of research undertaken by New Zealand government and fishing industry 74

3.3.2 Analysis of underwater video from Australian blue grenadier trawl fishery .. 75

3.4 Results and Discussion ... 75

3.4.1 Evidence that SLEDs allow sea lions to escape from trawl nets and that they survive… 75

3.4.2 Evidence of a low risk of significant head trauma when sea lions come into contact with stainless steel SLED grids. ... 78

3.4.3 Lack of evidence that dead pinnipeds passively drop out from top-opening escape holes ... 80

3.5 Summary and future directions ... 80

3.6 Appendix: Review and analysis of underwater video of seal interactions in blue grenadier trawl nets fitted with SEDs (full report in Hamilton et al. (2019)) ... 83

Chapter 4 — Current fisheries bycatch levels unlikely to be the main driver of New Zealand sea lion (*Phocarctos hookeri*) population trajectory at the Auckland Islands ... 87

4.1 Abstract ... 87

4.2 Introduction ... 88

4.3 Methods ... 95

Parameter values used in the modelling ... 95

Sensitivity analyses ... 99

4.4 Results ... 101

Base Model with varying levels of fisheries-related mortality ... 101

Base Model with varying frequency of epizootic mortality ... 102

Base Model with high frequency of epizootic mortality and varying levels of fisheries-based mortality ... 102

Sensitivity analyses ... 103

4.5 Discussion ... 106

Chapter 5 — Population growth of an endangered pinniped – the New Zealand sea lion (*Phocarctos hookeri*) – is limited more by high pup mortality than fisheries bycatch ... 110

5.1 Abstract ... 110

5.2 Introduction ... 111

5.3 Methods ... 117

Population size estimate for Campbell Island ... 117

Potential Biological Removal (PBR) ... 117

Population Viability Analysis (PVA) modelling ... 118

Assessing influence of reduced pup mortality ... 120

Assessing impact of fisheries bycatch........ 120
5.4 Results 123
Potential Biological Removal (PBR)........ 123
Population Viability Analysis (PVA)........ 124
5.5 Discussion........ 128
5.6 Appendices........ 132
Appendix 5A: Population size estimate for Campbell Island:........ 132
Appendix 5B: Potential Biological Removal (PBR)........ 132
Appendix 5C: Population Viability Analysis........ 134
Chapter 6 — General Discussion and Future Directions........ 139
Global review of technical mitigation to reduce marine mammal bycatch........ 139
Effective bycatch mitigation has reduced NZ sea lion mortality........ 141
Reassessment of NZ sea lion conservation status........ 144
Future directions........ 146
References........ 148

List of figures

Figure 3.1: The four current NZ sea lion breeding populations located at Campbell Island, the Auckland Islands, Stewart Island and the Otago coast of the South Island, NZ. (Figure published in Hamilton & Baker (2019)). 71

Figure 3.2: The observed number of captures (dark grey triangles) and mean estimated captures (black squares; error bars = 95% confidence interval) per year of NZ sea lions in squid trawl fisheries around the Auckland Islands from 1995/96 to 2012/13. The % observer coverage (light grey circles) is also shown. Data from 1995/96 to 2012/13 from Abraham et al. (2016); data from 2013/14 to 2015/16 from MPI (2018b). 72

Figure 3.3: Standard Sea Lion Exclusion Device used within the Auckland Islands squid trawl fishery (figure provided by Deepwater Group Ltd, New Zealand). 73

Figure 4.1: Total estimated pup production and the proportion of those recorded dead during annual mark/recapture estimate field work (late December to mid-February) for NZ sea lions at the Auckland Islands 1994/95–2016/17. Data sources: pre 2012/13 Chilvers (2012c); 2012/13–2016/17 Childerhouse et al. (2017); 2017/18 Childerhouse et al. (2018); 2018/19 Dodge et al. (2019). 89

Figure 4.2: For the Auckland Islands squid trawl fishery from 1995/96 to 2012/13: A) the observed number of captures (blue triangles) and mean estimated captures with 95% confidence intervals (black squares before SLEDs, yellow squares during SLED implementation and refinement, red squares with standardised SLEDs deployed), and B) the % observer coverage (blue open circles) and number of tows (black circles). Data for 1995/96 to 2012/13 from Abraham et al. (2016). 92

Figure 4.3: For 238 individuals tagged at the Auckland Islands that were re-sighted at a different location, the number seen at each location. Tag re-sight records for 1987–2016 from MPI (2018a). 99

Figure 4.4: Model sensitivity analyses showing the change in NZ sea lion population growth rate given different estimates of adult female annual mortality (top graph) and % female breeding participation (bottom graph). Sensitivity was undertaken using all other parameter values from the Base Model (Table 4.4 Model 1). 106

Figure 5.1:NZ sea lion pup count estimates for Campbell Island from 1985–2018. Empty Open circles indicate direct pup counts; dark grey circles indicate a mixture of mark-recapture and direct counts (noting that 1998 was an incomplete count, McNally et al. (2001)) with 95% confidence intervals from the mark-recapture component in 2003; and, from 2008 onwards, black circles indicate estimates using consistent mark-recapture methodology based on counts of tagged pups and untagged dead pups. Data from Childerhouse et al. (2005) (and references within), Childerhouse et al. (2015) and NZ Department of Conservation (unpublished data). 115

Figure 5.2: From 1996–2017, A) the annual observed number of captures (black triangles) and mean estimated captures (black open circles; error bars = 95% confidence interval) of NZ sea lions, and B) % observer coverage (black circles) and number of tows (open squares) for the Campbell Island SBW fishery. The red arrow indicates when Sea Lion Exclusion Devices (SLEDs) were deployed in this fishery following the 2013 season. Data from 1996–2016 from MPI (2018b), Abraham and Berkenbusch (2017) and, for 2017, NZ Ministry for Primary Industries (unpublished data). 116

Figure 5.3: From 2008–2018, using Base Model 1 parameters with a starting NZ sea lion population of 2,624 in 2008 (based on a pup count of 583 and pup multiplier of 4.5), the predictive model population trajectories and standard deviations with (A) and without (B)

immigration, compared with four population estimates (using a pup multiplier of 4.5) based on Campbell Island pup counts in 2008, 2010, 2015 and 2018 (see Figure 5.1). 127

Figure 6.1: Using available estimates from 2010/11–2018/19, the estimated pup production for NZ sea lions at A) Stewart Island (with 2018/19 considered an underestimate) and B) the Otago coast, South Island (note different scales on y-axis). Data from Roberts and Doonan (2016), MPI (2017b), Department of Conservation (DoC) (2018), Department of Conservation (DoC) (2019), Department of Conservation (DoC) (2020). Fitted lines represent simple linear regressions. 146

List of tables

Table 2.1: Summary of whether a technical measure developed to reduce pinniped and/or cetacean bycatch in commercial trawl, purse seine, longline, gillnet and pot/trap operations has been assessed (A) and if there is evidence that it is effective (E) in reducing bycatch. 11

Table 2.2: Subjective evaluation of the economic viability, practicality, impact on target catch and ease of compliance monitoring for mitigation measures shown to be, or have the potential to be, effective in reducing pinniped and/or cetacean bycatch in trawl, purse seine, longline, gillnet and pot and trap fishing gear. Note that, although this evaluation provides a general overview, due to fishery-specific characteristics (e.g. size of target species, operational elements), responses may differ across fisheries. 12

Table 4.1: NZ sea lion capture data from New Zealand trawl fisheries from 2006 to 2013. Data from Abraham et al. (2016). 93

Table 4.2: NZ sea lion demographic parameters, bycatch levels and mass mortality disease levels used in the Population Viability Assessment (PVA) for the Auckland Islands NZ sea lion population. Further details of how parameter values were derived are in Methods. . 100

Table 4.3: NZ sea lion dispersal data from Otago, NZ (columns 1–3) used to estimate the number of emigrant individuals per age class for males and females (columns 5 and 6) in the Population Viability Analysis (PVA) models for the Auckland Islands. Data in column 2 is the age distribution of 40 new identified males (not including pups) sighted from 1995–1999 in Otago from Table 1 in McConkey et al. (2002a). The proportions in column 3, based on the data in column 2, were used to determine the age distributions of males and females in columns 5 and 6. 101

Table 4.4: Model predictions for the Auckland Islands NZ sea lion population showing mean stochastic population growth rate (stochastic r), mean final population size (N) and standard deviation (SD) after 30 years. Each model was run for a 30-year period with 2000 simulations. 104

Table 5.1: NZ sea lion demographic parameters and bycatch levels used in the Population Viability Analysis (PVA) for the Campbell Island NZ sea lion population. 122

Table 5.2: Potential Biological Removal estimates (PBRs) for the New Zealand sea lion population on Campbell Island using a minimum population size of 2 793 and a range of recovery factor (F_r,) and maximum theoretical net productivity rate (R_{max}) values. 125

Table 5.3: Model predictions for the NZ sea lion population on Campbell Island showing mean stochastic population growth rate (r, with standard deviation, SD; estimates rounded to 3 decimal places) and mean final population size (N, with SD; estimates rounded to nearest 10) after 30 years. 126

Thesis Abstract

Marine mammals, particularly cetaceans and pinnipeds, are killed as incidental bycatch in fisheries around the world. Management measures to reduce this bycatch and improve the conservation outcomes of affected bycatch species include the implementation of temporal and spatial fishery closures, operational protocols and the use of technical mitigation measures, such as the installation of devices or adjustments to operational gear.

In my thesis, I first undertake a global review and assessment of technical mitigation measures used to reduce the bycatch of marine mammals, particularly cetaceans and pinnipeds, in commercial trawl, purse seine, longline, gillnet and pot or trap fishing gear. For some gear types and taxa, there are currently limited technical options that show strong evidence for effectively reducing bycatch. Research and development is urgently needed into effective measures to reduce small cetacean bycatch in trawl nets, the bycatch of some small cetacean and pinniped species in gillnets and the entanglement of large baleen whales in pot or trap buoy-lines. While there are promising results from options such as loud pingers to deter dolphins from trawls and rope-less pot or trap systems, continued research and development in these areas should be a high priority. Few technical mitigation measures have undergone robust testing to determine their effectiveness in reducing mortality of the bycatch species while maintaining operational efficiency and target catch quality and quantity. Examples of effective measures include acoustic devices (pingers) that have reduced the bycatch of some small cetacean species, particularly harbour porpoise *Phocoena phocoena*, in gillnets, and appropriately designed exclusion devices which have reduced pinniped bycatch in some trawl fisheries.

As a case study, my thesis focused on a ‘Sea Lion Exclusion Device’, or SLED, developed to reduce bycatch of the endangered New Zealand (NZ) sea lion (*Phocarctos hookeri*) in trawl fisheries. The SLED, installed before the trawl codend, has a stainless steel upwardly inclined grid that directs entrapped sea lions to a top-opening escape hole covered with a backward-facing hood. Target species are able to pass through the grid and into the codend. Research, development and implementation of the SLED was undertaken to reduce NZ sea lion bycatch in the squid trawl fishery operating near the Auckland Islands in sub-Antarctic New Zealand. Observed bycatch has been greatly reduced following SLED implementation. However, there has been scepticism and concern that significant ‘cryptic’ or unaccounted mortality may be

occurring, particularly sea lions actively exiting the net but exceeding their breath-holding capabilities before they reach the surface.

I assessed the effectiveness of SLEDs by reviewing and evaluating trials and tests of sea lion interactions with SLEDs in trawl nets. I complemented this assessment by developing and fitting population models for the two largest NZ sea lion populations on the Auckland Islands and Campbell Island to evaluate the impact of fisheries bycatch on population growth, particularly after bycatch mitigation implementation.

The available evidence shows SLEDs are effective in reducing NZ sea lion bycatch, sea lions are able to escape via SLEDs and are unlikely to sustain life-threatening injuries, and SLEDs contribute to reduced rates of observed sea lion mortality in trawl fisheries. Sea lion carcasses are also unlikely to passively drop out from a SLED top-opening escape hole.

Further compelling evidence that cryptic mortality is unlikely to be significant is provided through NZ sea lion demographic parameters, population viability assessments and current population trajectories. Modelling of both the Auckland Islands and Campbell Island populations showed that current bycatch estimates from relevant trawl fisheries are sustainable following effective bycatch mitigation, and population growth rates are positive although slow. Modelling also indicated that disease events causing reduced pup production may greatly impact the population growth at the Auckland Islands, and management actions that reduce pup mortality would lead to increased growth rates for both populations.

The development, refinement and testing of the SLED is an example of effective mitigation resulting in encouraging bycatch reduction and conservation outcomes. The conservation of NZ sea lions over the past decade should be regarded as a ‘good news’ story, with fisheries bycatch effectively mitigated, the population decline observed at the Auckland Islands halted, all other breeding populations increasing or stable, and the species breeding range expanded. While effective bycatch mitigation implementation has significantly improved the conservation status of the NZ sea lion, the focus should remain on addressing other threats, such as disease, which appear to be slowing population recovery.

Chapter 1 — General Introduction

Marine mammals, particularly cetaceans and pinnipeds, are killed in fishing operations around the world. Cetaceans and pinnipeds interact with fisheries as they may feed on the same target species or associated non-target species of that fishery, be attracted to fishing operation discards, and/or passively encounter fishing gear in the water column (Fertl and Leatherwood 1997; Hamer et al. 2012; Morizur et al. 1999). These interactions may result in the mortality of individuals that become entangled or caught by nets, lines or hooks. Bycatch (assumed to equate to mortality) in active fishing gear or entanglement in the supporting elements of gear (e.g., buoylines) are identified as key threats to marine mammal populations (Jaiteh et al. 2013; Read 2008; Reeves et al. 2013).

Bycatch in trawl (Hamer and Goldsworthy 2006; Robertson and Chilvers 2011; Thompson et al. 2013a; Thompson et al. 2013b), purse seine (Hall 1998; Hamer et al. 2008), longline (Hamer et al. 2012; Werner et al. 2015), gillnet (Hamer et al. 2013; Reeves et al. 2013; Slooten and Dawson 2010) and pot or trap (Johnson et al. 2005; Vanderlaan et al. 2011) fisheries have been identified as a major threat to many cetacean and pinniped species. Other types of gear, such as those used in troll and squid jigging fisheries, are considered more selective in targeting species and, therefore, have less non-target bycatch risk (Wakefield et al. 2017). Addressing bycatch issues requires an understanding of the gear and operational specifications for each fishing type.

Trawl operations tow a net through the water to capture target species. Marine mammal mortality occurs when individuals enter the net and become trapped (Du Fresne et al. 2007; Hamer and Goldsworthy 2006). Marine mammals are more frequently caught in pelagic or midwater trawls compared to demersal (bottom) trawls, possibly due to midwater trawls targeting the same pelagic species as those eaten by marine mammals, being towed at relatively high speeds, having larger nets than most demersal trawls, and usually operating within marine mammals diving ranges and for extended periods, which may exceed their breath-holding capacities (Fertl and Leatherwood 1997; Hall et al. 2000). Small cetaceans, seals and sea lions are also known to deliberately enter nets to feed on captured fish (Fertl and Leatherwood 1997; Hamer and Goldsworthy 2006; Lyle et al. 2016; Wakefield et al. 2017).

Purse seine operations encircle a fish shoal with a large circular ‘wall’ of net. Typically, the top of the net is attached to a buoyed float line and the bottom has steel wire threaded through rings which can be ‘pursed’ together to capture and stop fish escaping downwards under the net (AFMA 2017; Hall and Roman 2013). Encircled marine mammals may be able to escape before net pursing is complete, either by diving under the net or through the opening before the net ends are fully together, but there are no clear escape options once pursing is complete (Hall and Roman 2013).

Longline operations target fish species using a series of baited hooks on a line set either on the seabed (demersal longlining), floated off the bottom at various fishing depths (semi-pelagic longlining) or suspended from floats at the surface (pelagic longlining). Marine mammals are often attracted to longline operations as a source of food, particularly if hooked fish are accessible and easy to take (Ashford et al. 1996). Consequently, most marine mammal interactions result in depredation (i.e. damage or removal) of the target catch, although marine mammal bycatch risk from these interactions can an issue.

Gillnets passively catch target species using a wall of virtually invisible netting typically constructed of monofilament or multifilament nylon. Mesh sizes are designed for the head of target fish to pass through and become entangled, usually around the gills (He 2006b). Gillnets incidentally entangle a wide range of marine mammals and other non-target taxa which may drown when caught in set gillnets (anchored at each end with bottom weights and floats at the top) or, if entangled in drift gillnets (held at the required fishing depth using a system of weights and buoys), may drag gear with them which can impact feeding ability, constrict growth and/or cause infection. Marine mammal interactions may also involve associated lines as well as the net itself (He 2006b).

Pots and traps are enclosures that trap or entangle fish, crustaceans or molluscs by enticing them inside, either using bait or because the enclosure appears to provide a refuge. Depending on the target species, most pots and traps have one or more entrances on the top or sides, are set on the bottom, and have a haul-in line, surface float or dan buoy to mark their position. Bycatch occurs when marine mammals, primarily pinnipeds and small cetaceans, become trapped when they attempt to feed on captured target species within the pot/trap (Campbell et al. 2008; Konigson et al. 2015), or when animals, particularly large baleen whales, become entangled in the associated lines (Benjamins et al. 2012; Johnson et al. 2005; Kraus et al. 2016; Reeves et al. 2013; Thomas et al. 2016).

Over the past three decades or so, a number of bycatch mitigation strategies and techniques have been developed and designed to reduce the bycatch mortality of marine mammals in commercial fishing operations. The biological and behavioural characteristics of target and non-target species, temporal and spatial overlap with fishing activities, and fishing gear type and configuration (e.g. pelagic or demersal) may all influence the vulnerability of a species to bycatch (Brown et al. 2013; Couperus 1997; Woodley and Lavigne 1991) and require consideration in the research and development of fisheries and species-specific bycatch mitigation. In some fisheries, large amounts of resources have been invested in developing and refining gear modifications, particularly devices, that aim to reduce non-target species bycatch. For the first part of my thesis, I undertake a global review and assessment of all known available technical mitigation that has been developed to reduce marine mammal bycatch in commercial trawl, longline, purse seine, gillnet, and pot/trap fishing gear.

As a case study, I then focus on the endangered New Zealand (NZ) sea lion (*Phocarctos hookeri)*, which is killed in midwater trawl fisheries operating around southern and sub-Antarctic New Zealand. NZ sea lions were up-listed to *Endangered* on the IUCN Red List (Chilvers 2015), although they have been recently down-listed from *National Critical* to *Nationally Vulnerable* on the New Zealand Threat Classification system (Baker et al. 2019). New Zealand sea lions are endemic to NZ with contemporary breeding populations at four main locations; the Auckland Islands, Campbell Island, Stewart Island, and the Otago coast (including the Otago Peninsula and the Catlins) (see Figure 3.1). The trawl fishery that has recorded the highest level of NZ sea lion bycatch is the Auckland Islands squid trawl fishery (targeting *Nototodarus sloanii*), although bycatch has also been recorded in the Auckland Islands scampi (*Metanephrops challenger*) fishery, Auckland Islands non-squid/scampi target fisheries, the Campbell Island southern blue whiting (*Micromesistius australis*) fishery and Stewart-Snares shelf trawl fisheries (Thompson et al. 2013b).

To address the NZ sea lion bycatch issue, several mitigation measures have been implemented. In particular, a ‘Sea Lion Exclusion Device’ (SLED) was developed to reduce NZ sea lion bycatch in the Auckland Islands squid trawl fishery operating near the sea lions’ largest breeding population on the Auckland Islands (MAF 2012). A sea lion within a trawl net encounters the hard steel grid of the SLED and is directed upwards to a top-opening escape hole where it can swim out. Target species, such as squid, pass through the grid and are retained in the codend of the trawl net. These SLED devices have been deployed in the Auckland Islands squid trawl fishery since 2004 and in the Campbell Island southern blue whiting trawl fishery since 2014, and have greatly reduced observed bycatch levels in both these fisheries (Hamilton and Baker 2016a; Hamilton and Baker 2019).

However, interactions with all types of active fishing gear potentially result in a level of cryptic or unaccounted mortality of marine mammals (Uhlmann and Broadhurst 2015; Warden and Murray 2011) which may include injured or dead animals dropping out of (or off) gear (Hamer et al. 2011; Kindt-Larsen et al. 2012; Lyle et al. 2016) as well as post-escape mortality resulting from injuries or stress. Cryptic mortality is poorly understood and rarely quantified for marine mammals in commercial fishing gear. Poor knowledge of cryptic mortality restricts understanding of the impacts of fisheries on marine mammal populations and, without assessment of cryptic mortality, fisheries-related mortality may be under-estimated. Ensuring a fishery has reduced its direct impacts on bycatch species requires verification of mitigation effectiveness in reducing observed bycatch to sustainable levels as well as the likelihood of survival following interactions with gear.

While a focus on mitigation development and implementation that effectively reduces bycatch mortality rates is crucial, there is often a lack of assessment of the actual impact of this mortality on the bycatch species population. Assessing the impact of fisheries bycatch requires more than just an understanding of the level of bycatch of a particular species in a fishery. Understanding the impact of fisheries interactions at the population level is important to enable prioritization of management actions. In an environment of limited resources and the need to prioritise funding, absolute bycatch numbers (e.g. dead animals per year) are not, in themselves, a cause for action and assessments of the actual impact of a threat to the species in question are needed. In this way, resources can be focussed on areas and problems that are the highest priority.

A useful approach to assessing the impact of fisheries is to estimate key demographic rates (survival, productivity, immigration, and emigration), and the relative contribution they make to population growth rate and population trends. If bycatch levels and other forms of mortality are unsustainable, populations will have negative population growth. However, robust analysis is dependent on the availability of demographic data, ideally collected over multiple consecutive years, to allow calculation of population trends.

As a component of assessing bycatch sustainability and evidence of cryptic mortality, I undertook population modelling of the Auckland Islands and Campbell Island populations of NZ sea lions, which have been most subject to trawl fisheries bycatch. Published literature and analysis of existing long-term demographic data sets for NZ sea lions were incorporated into a Population Viability Analysis (PVA) for each population to assess the conservation implications of fishing-related mortality (particularly following implementation of mitigation) and other threats, and provide guidance for future management. Assessment of the Campbell Island population also considered Potential Biological Removal (PBR) levels (Wade 1998) as part of evaluating the ecological sustainability of relevant fisheries.

In summary, my thesis is organized into four main chapters (Chapters 2–5). Each chapter is designed to stand-alone and, as all have been published in peer-reviewed journals, they retain this style and layout. However, all references are collated in one section at the end of the thesis. A brief outline of my thesis is as follows:

- In Chapter 2*,* I undertake a global review of technical mitigation that has been developed, tested, and/or implemented to reduce 'bycatch' in fisheries. More specifically, I review and assess the effectiveness of mitigation in reducing cetacean and pinniped bycatch in commercial trawl, purse seine, longline, gillnet and pot/trap fishing operations.

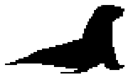
- In Chapter 3, I focus on my case study, the mitigation of NZ sea lion bycatch in midwater trawl fisheries. I assess whether NZ sea lions can escape trawl nets via SLEDs, the likelihood of them incurring a life-threatening injury during this interaction and whether dead pinnipeds are likely to 'drop out' of an exclusion device's top-opening escape hole.

- In Chapter 4, I use data from long-term studies of NZ sea lion breeding biology to undertake population modelling of the largest NZ sea lion population on the Auckland Islands and assess the sustainability of bycatch rates following the implementation of effective mitigation techniques.

- In Chapter 5, I develop models for the Campbell Island population of NZ sea lions and assess the sustainability of fisheries bycatch levels following the implementation of SLEDs in the southern blue whiting trawl fishery using both the PVA and PBR techniques. I also assess the demographic parameters with the greatest influence on sea lion population trajectory.

I conclude my thesis with a ‘General Discussion and Future Directions’ chapter (Chapter 6) where I summarise the primary findings from each chapter and discuss them in the context of their implications for NZ sea lion conservation and population status.

Chapter 2 — Technical mitigation to reduce marine mammal bycatch and entanglement in commercial fishing gear: lessons learnt and future directions



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***Note that the 13th meeting of the Conference of the Parties to the UNEP Convention on Migratory Species (COP 13, Gandhinagar, 2020) adopted the following Decision 13.61 “Parties that manage fisheries with identified marine mammal bycatch problems are urged to take account of the Hamilton and Baker (2019) assessment, available as UNEP/CMS/COP13/Inf.11”* [*https://www.cms.int/sites/default/files/document/cms_cop13_decisions_e_rev.1.pdf*](https://www.cms.int/sites/default/files/document/cms_cop13_decisions_e_rev.1.pdf)

2.1 Abstract

Fisheries bycatch is one of the biggest threats to marine mammal populations. A literature review was undertaken to provide a comprehensive assessment and synopsis of gear modifications and technical devices to reduce marine mammal bycatch in commercial trawl, purse seine, longline, gillnet and pot/trap fisheries. Successfully implemented mitigation measures include acoustic deterrent devices (pingers) which reduced the bycatch of some small cetacean species in gillnets, appropriately designed exclusion devices which reduced pinniped bycatch in some trawl fisheries, and various pot/trap guard designs that reduced marine mammal entrapment. However, substantial development and research of mitigation options is required to address the bycatch of a range of species in many fisheries. No reliably effective technical solutions to reduce small cetacean bycatch in trawl nets are available, although loud pingers have shown potential. There are currently no technical options that effectively reduce marine mammal interactions in longline fisheries, although development of catch and hook protection devices is promising. Solutions are also needed for species, particularly pinnipeds and small cetaceans, that are not deterred by pingers and continue to be caught in static gillnets. Large whale entanglements in static gear, particularly buoy lines for pots/traps, needs urgent

attention although there is encouraging research on rope-less pot/trap systems and identification of rope colours that are more detectable to whale species. Future mitigation development and deployment requires rigorous scientific testing to determine if significant bycatch reduction has been achieved, as well as consideration of potentially conflicting mitigation outcomes if multiple species are impacted by a fishery.

2.2 Introduction

Marine mammals are incidentally killed in a range of fisheries throughout the world (Lewison et al. 2014; Read et al. 2006). This *bycatch* in active fishing gear is one of the biggest threats to marine mammal populations, particularly cetaceans (whales, dolphins and porpoises) and pinnipeds (e.g. seals and sea lions) (Jaiteh et al. 2013; Read 2008; Reeves et al. 2013). As these species are long-lived with high adult survival and low breeding productivity, populations are often slow to recover from declines, even under conducive environmental conditions. Therefore, anthropogenic activities that increase mortality levels, such as fisheries bycatch, can have significant, long-term population impacts (Gilman 2011; Lewison et al. 2004; Reeves et al. 2003).

Cetaceans and pinnipeds interact with fisheries as they may: 1. feed on the same target species or associated non-target species of a fishery, 2. be attracted to fishing operation discards, and/or 3. passively encounter fishing gear in the water column (Fertl and Leatherwood 1997; Hamer et al. 2012). These interactions may result in the bycatch of individuals caught in active fishing components (e.g. nets, hooks, traps), or entangled in supporting gear and lines. Bycatch in trawl, purse seine, longline, gillnet and pot/trap fisheries has been identified as a major threat to many species (Hall 1998; Hamer et al. 2012; Hamer and Goldsworthy 2006; Hamer et al. 2008; Knowlton et al. 2012; Reeves et al. 2013; Werner et al. 2015). Other gear types, such as those used in troll and squid jigging fisheries, are considered to be more selective in targeting species and, therefore, have less bycatch risk (Wakefield et al. 2017).

Over the past decade, there has been heightened awareness and attention on the development of solutions to reduce fisheries bycatch. For example, the Food and Agriculture Organization of the United Nations (FAO), as part of an ongoing commitment to bycatch management work, convened a workshop to consider means to reduce marine mammal mortality in fisheries and aquaculture operations (FAO 2018). Also, a number of bycatch mitigation reviews have focussed on particular aspects of mitigation or gear type, or on certain species or species groups (Dawson et al. 2013; Geijer and Read 2013; Hamer et al. 2012; How et al. 2015; Laverick et al. 2017; Leaper and Calderan 2018; Werner et al. 2006; Werner et al. 2015). However, there is no readily accessible synthesis of best practice mitigation methods for marine mammals and, furthermore, the high level of bycatch that continues to occur in fisheries around the world (Gray and Kennelly 2018; Reeves et al. 2013) necessitates an update and expansion from previously published assessments. This chapter presents the first comprehensive global review of technical mitigation measures designed to reduce marine mammal bycatch in commercial fishing gear, including assessments of mitigation testing, effectiveness and, where relevant, operational deployment, and a synthesis of best practice mitigation and areas requiring greater attention.

2.3 Methods and scope

Although there has been considerable progress in some fisheries regarding the development, testing and implementation of mitigation measures to reduce marine mammal bycatch in commercial fishing gear, much of this information is not easily accessible. A literature review was undertaken using a range of sources including peer-reviewed journals, unpublished reports, magazine articles, conference papers, websites, and information from government and non-government organisations. An electronic literature search was conducted up to and including August 2018 using Web of Science and Google Scholar. Search terms were *bycatch*, *by-catch* and/or *mitigat** combined with: *fisher**, *trawl*, *purse seine*, *longline*, *gillnet*, *pot*, *trap*, *line*, *cetacean*, *whale*, *dolphin*, *porpoise*, *pinniped*, *seal*, *sea lion* in any field. References from other published papers and the authors' personal bibliographic resources were used to identify relevant papers. Key researchers were contacted via email or ResearchGate (https://www.researchgate.net/) to access relevant non-published reports.

Studies on the development and implementation of technical mitigation measures (i.e. gear modifications and mitigation devices) for marine mammal bycatch in commercial trawl, purse seine, longline, gillnet and pot/trap fishing gear were reviewed. Fisheries not considered to be high risk to marine mammal species, such as trolling and jigging (Arnould et al. 2003), and mitigation of mortalities from lost, discarded or abandoned gear (i.e. ghost fishing) were not included. Reviewed studies predominantly addressed cetacean and/or pinniped bycatch as most mitigation research has focussed on these taxa.

Technical measures are presented on a fishing gear basis (trawl, purse seine, longline, gillnet and pot/trap) with the exception of *pingers* and a range of *weakened gear*, which are applicable to different fishing gears and are therefore more effectively dealt with in a collated section. For each measure, the scientific evidence for mitigation effectiveness, caveats or uncertainties in the methods or results, research requirements and, where possible, recommendations for effective operational implementation were identified.

Although outside the scope of this review, it was apparent that effective bycatch mitigation strategies often comprise a suite of management measures in conjunction with technical mitigation. These include traditional input and output controls, operational adjustments through ‘codes of practice’ protocols (e.g. ‘move-on’ provisions, handling and release protocols) and implementation of appropriately designated spatial and/or temporal closures (Hamer and Goldsworthy 2006; Hamer et al. 2008; Hamer et al. 2011; Read 2013; Reyes et al. 2012; Rojas-Bracho and Reeves 2013; Slooten 2013; Tixier et al. 2014; Werner et al. 2015). Instigation of multi-jurisdictional agreements, regulations and/or legislation to facilitate mitigation implementation are also likely to be important (Geijer and Read 2013; Leaper and Calderan 2018).

2.4 Results of reviewed technical mitigation measure

A synopsis of the technical mitigation assessment is provided below, with details on mitigation and fishery-specific studies provided in Appendix 2A—2E. A summary of the assessment and effectiveness of each technical measure identified is provided in Table 2.1. Where appropriate, a subjective evaluation of the economic viability, practicality, impact on target catch and the ease of compliance monitoring for each technical measure is provided in Table 2.2. However, although this provides a general overview, due to fishery-specific characteristics (e.g. size of target species, operational elements), the evaluation responses are not definitive, and results may differ across fisheries. For example, a range of fishery-specific factors would affect the economic feasibility of mitigation implementation such as operational specifications, target species value and how much the mitigation reduces target species damage or depredation by bycatch species.

Table 2.1: Summary of whether a technical measure developed to reduce pinniped and/or cetacean bycatch in commercial trawl, purse seine, longline, gillnet and pot/trap operations has been assessed (A) and if there is evidence that it is effective (E) in reducing bycatch.

	TRAWL				**PURSE SEINE**				**LONGLINE**				**GILLNET**				**POT/TRAP**			
	Pinniped		**Cetacean**		**Pinniped**		**Cetacean**		**Pinniped**		**Cetacean**		**Pinniped**		**Cetacean**		**Pinniped**		**Cetacean**	
Technical measure	**A**	**E**	**A**	**E**	**A**	**E**	**A**	**E**	**A**	**E**	**A**	**E**	**A**	**E**	**A**	**E**	**A**	**E**	**A**	**E**
Acoustic Deterrent Devices (Pingers)	no	?	yes	?	—	—	—	—	no	?	yes	no	no	?	yes	yes	no	?	yes	?
Acoustic scarers; e.g. alarm or predator calls, explosions	—	—	—	—	?	no	yes	no	no	?	yes	no	—	—	—	—	—	—	—	—
Acoustically reflective nets	—	—	—	—	—	—	—	—	—	—	—	—	no	?	yes	no	—	—	—	—
Auto-trawl systems	no	?	no	?	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Back-down manoeuvre with Medina panels	—	—	—	—	no	?	yes	yes	—	—	—	—	—	—	—	—	—	—	—	—
“Buoyless” nets	—	—	—	—	—	—	—	—	—	—	—	—	no	?	no	?	—	—	—	—
Catch protection devices – demersal longline	—	—	—	—	—	—	—	—	no	?	yes	?	—	—	—	—	—	—	—	—
Catch protection devices (triggered) – pelagic longline	—	—	—	—	—	—	—	—	no	?	yes	?	—	—	—	—	—	—	—	—
“Dolphin gate” with additional weights	—	—	—	—	no	?	yes	?	—	—	—	—	—	—	—	—	—	—	—	—
Exclusion device: hard grid & top-opening escape	yes	yes	yes	no	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Exclusion device: soft/flexible grid & top-opening escape	yes	?	yes	no	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Exclusion device: hard grid & bottom-opening escape	yes	?	yes	?	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Exclusion device: soft/flexible grid & bottom-opening escape	no	?	yes	no	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Mesh enlargement	—	—	—	—	no	no	?	no	—	—	—	—	—	—	—	—	—	—	—	—
Net binding	no	?	no	?	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Net colour	no	?	no	?	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Passive acoustic deterrents	—	—	—	—	—	—	—	—	no	?	yes	?	—	—	—	—	—	—	—	—
Pot/trap excluder devices	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	yes	yes	yes	yes
Reduced strength rope	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	no	?	no	?
Reduced strength nets	—	—	—	—	—	—	—	—	—	—	—	—	yes	?	no	?	—	—	—	—
Rope colour changes	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	yes	?
Rope or mesh barriers	no	?	yes	?	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Rope-less pot/trap systems	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	yes	?
“Seal socks”	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	yes	yes	no	?
Sinking groundlines	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	yes	no
Stiff ropes	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	yes	no
Visually detectable nets	—	—	—	—	—	—	—	—	—	—	—	—	no	?	no	?	—	—	—	—
Weak hooks	—	—	—	—	—	—	—	—	no	?	yes	?	—	—	—	—	—	—	—	—
Weak links	—	—	—	—	—	—	—	—	—	—	—	—	no	?	no	?	no	?	yes	?

“?” for ***assessed*** category = unclear whether there has been any assessment of the measure.
“?” for ***effective*** category = lack of knowledge of the measure’s effectiveness, results have been inconclusive and/or more trials are needed.
“—” = measure is not applicable for relevant fishing gear.

Table 2.2: Subjective evaluation of the economic viability, practicality, impact on target catch and ease of compliance monitoring for mitigation measures shown to be, or have the potential to be, effective in reducing pinniped and/or cetacean bycatch in trawl, purse seine, longline, gillnet and pot and trap fishing gear. *Note that, although this evaluation provides a general overview, due to fishery-specific characteristics (e.g. size of target species, operational elements), responses may differ across fisheries.*

	TRAWL				**PURSE SEINE**				**LONGLINE**				**GILLNET**				**POT/TRAP**			
Technical measure	**EV**	**P**	**ITC**	**CR**	**EV**	**P**	**ITC**	**CR**	**EV**	**P**	**ITC**	**CR**	**EV**	**P**	**ITC**	**CV**	**EV**	**P**	**ITC**	**CR**
Acoustic Deterrent Devices (Pingers)	yes	maybe	no	OBS	—	—	—	—	no	no	unk	OBS	yes	yes	no	OBS	yes	yes	no	OBS
Acoustic scarers; e.g. alarm or predator calls, explosions	—	—	—	—	—	—	—	—	maybe	maybe	unk	OBS	—	—	—	—	—	—	—	—
Acoustically reflective nets	—	—	—	—	—	—	—	—	—	—	—	—	maybe	yes	no	DCK*	—	—	—	—
Auto-trawl systems	yes	yes	no	OBS	—	—	—	—	—	—	—	—	—	—			—	—	—	—
Back-down manoeuvre with Medina panels	—	—	—	—	yes	yes	no	OBS	—	—	—	—	—	—	—	—	—	—	—	—
“Buoyless” nets	—	—	—	—	—	—	—	—	—	—	—	—	yes	maybe	maybe	OBS	—	—	—	—
Catch protection devices – demersal longline	—	—	—	—	—	—	—	—	yes	maybe	no	OBS	—	—	—	—	—	—	—	—
Catch protection devices (triggered) – pelagic longline	—	—	—	—	—	—	—	—	yes	maybe	no	OBS	—	—	—	—	—	—	—	—
“Dolphin gate” with additional weights	—	—	—	—	yes	maybe	no	OBS	—	—	—	—	—	—	—	—	—	—	—	—
Exclusion device: hard grid & top-opening escape	yes	maybe	maybe	OBS	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Exclusion device: soft/flexible grid & top-opening escape	yes	maybe	maybe	OBS	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Exclusion device: hard grid & bottom-opening escape	yes	maybe	maybe	OBS	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Exclusion device: soft/flexible grid & bottom-opening escape	yes	maybe	maybe	OBS	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Net binding	yes	yes	no	OBS	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Net colour	maybe	yes	no	DCK§	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

	TRAWL				PURSE SEINE				LONGLINE				GILLNET				POT/TRAP			
Technical measure	**EV**	**P**	**ITC**	**CR**	**EV**	**P**	**ITC**	**CR**	**EV**	**P**	**ITC**	**CR**	**EV**	**P**	**ITC**	**CV**	**EV**	**P**	**ITC**	**CR**
Passive acoustic deterrents	—	—	—	—	—	—	—	—	yes	maybe	no	OBS	—	—	—	—	—	—	—	—
Pot/trap excluder devices	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	yes	yes	maybe	OBS
Reduced strength rope	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	yes	yes	no	DCK
Reduced strength nets	—	—	—	—	—	—	—	—	—	—	—	—	yes	yes	unk	DCK	—	—	—	—
Rope colour changes	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	yes	yes	no	DCK
Rope or mesh barriers	yes	maybe	maybe	OBS	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Rope-less pot/trap systems	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	yes	maybe	no	DCK
“Seal socks”	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	yes	yes	unk	OBS
Visually detectable nets	—	—	—	—	—	—	—	—	—	—	—	—	yes	yes	maybe	DCK	—	—	—	—
Weak hooks	—	—	—	—	—	—	—	—	maybe	maybe	no	DCK	—	—	—	—	—	—	—	—
Weak links	—	—	—	—	—	—	—	—	—	—	—	—	yes	yes	unk	OBS	yes	yes	no	OBS

Economically viable: based on the cost for initial outlay plus any ongoing maintenance = *yes*, *no*, *maybe*;

Practical for use: i.e. has no great impact on fishing operation and operational efficiency = *yes*, *no*, *maybe*;

Impact target catch: i.e. could cause a reduction in the amount or quality of catch = *yes*, *no*, *maybe*, *unk* (= *unknown*); and,

C***ompliance requirement:*** either requiring at-sea observations (*OBS*) or whether dockside inspections would be adequate (*DCK*).

“—” = measure is not applicable for relevant fishing gear or assessed to be ineffective for pinnipeds and cetaceans in that type of gear (see Table 1).

*Only if all nets on board were of appropriate material. A mix of netting material would require ‘OBS’.

§Only if all trawl nets on board were of appropriate colour.

2.4.1 Mitigation relevant to multiple types of fishing gear

Pingers (Acoustic Deterrent Devices)

Pingers, small electronic devices with relatively low acoustic outputs (<160 dB), were developed to reduce high levels of small cetacean bycatch in gillnets (Dawson et al. 2013; Kraus et al. 1997; Reeves et al. 2013). Pingers also include louder devices (>132 dB) to deter marine mammals from trawl nets or to reduce pinniped or odontocete interactions and depredation around aquaculture, longline or pot/trap operations (Dawson et al. 2013; Hamer et al. 2012; Mackay and Knuckey 2013). The effectiveness of pingers in reducing bycatch differs between trawl, longline, gillnet and pot/trap gear (Table 2.1), and between species and fisheries. Furthermore, the economic viability of deploying pingers varies between gear types. It is likely to be more economically viable to deploy pingers on gear contained within a relatively small range (e.g. gillnets, trawls, pot/trap lines) than using pingers to deter marine mammals from longlines, which can extend over tens of kilometres (Table 2.2).

For trawl fisheries (Appendix 2A), while there are likely to be inter- and intra-specific differences in responses to pingers with different signals, the effectiveness of pingers in reducing cetacean bycatch is unclear. Correctly deployed, loud pingers (e.g. *Dolphin Dissuasive Devices*®, *'DDD'*) may reduce common dolphin (*Delphinus delphis*) bycatch in seabass (*Dicentrarchus labrax*) pair trawl fisheries (Northridge et al. 2011), although decreases in reported bycatch may be partly due to reduced fishing effort (de Boer et al. 2012) and results from other trials (with different pinger models) were inconclusive (Morizur et al. 2008). Furthermore, controlled experiments in the absence of the loud operational conditions of trawls indicated pingers may not provide a consistently effective deterrent for common dolphins (Berrow et al. 2009). Pingers may also have less effect on foraging compared to travelling groups of cetaceans (van Marlen 2007). Neither DDDs nor quieter pingers were effective in reducing bottlenose dolphin (*Tursiops truncatus*) interactions in Australia's Pilbara demersal fish trawl fishery (Santana-Garcon et al. 2018; Stephenson and Wells 2006). While one study suggested pingers may increase rates of bottlenose and Risso's (*Grampus griseus*) dolphin bycatch in mid-Atlantic bottom trawl fisheries, there is low confidence in this finding due to small sample sizes and limited information on the type and quantity of deployed pingers (Lyssikatos 2015).

In longline fisheries (Appendix 2C), while there has been a high degree of variability in device design and deployment, there is no clear evidence that pingers effectively deter marine mammals (Hamer et al. 2012; Tixier et al. 2015; Werner et al. 2015). This may be largely due to the difficulty in protecting longlines which are set over large distances (Rabearisoa et al. 2012).

In gillnet fisheries (Appendix 2D), although pingers have effectively reduced the bycatch of some small cetacean species, the results are not universal and mitigation effectiveness is likely to be species- and fishery-specific. A number of studies have shown that pingers reduced harbour porpoise (*Phocoena phocoena*) bycatch (Dawson et al. 2013; Kraus et al. 1997; Larsen and Eigaard 2014; Larsen et al. 2013; Palka et al. 2008; Reeves et al. 2013). However, results for bottlenose dolphins have been less clear with some research reporting significantly reduced interactions (Crosby et al. 2013; Gazo et al. 2008; Leeney et al. 2007; Mangel et al. 2013), while others showed no deterrent effect (Cox et al. 2003; Erbe et al. 2016). Pingers have been ineffective, or the results have been inconclusive, in deterring Hector's dolphin (*Cephalorhynchus hectori*), tucuxi (*Sotalia fluviatilis*), and other small coastal species such as the Australian snubfin (*Orcaella heinsohni*) and humpback dolphin (*Sousa chinensis*) (Berg Soto et al. 2013; Dawson and Lusseau 2005; Dawson and Slooten 2005). Pingers may also attract some species, particularly pinnipeds, to depredate captured fish (Bordino et al. 2002; Mackay and Knuckey 2013). Although initial testing showed California sea lion (*Zalophus californianus*) and northern elephant seal (*Mirounga angustirostris*) bycatch reduced with pinger use (Barlow and Cameron 2003), monitoring of pinger deployment over 14 years subsequently showed sets with pingers had almost twice the amount of California sea lion bycatch although this increase was most likely due to increased sea lion abundance and was not considered to be caused by pinger use (Carretta and Barlow 2011). There is no indication pingers would reduce bycatch risk for other species of seal, sea lion or dugong (*Dugong dugon*) in gillnets (Bordino et al. 2002; Gearin et al. 2000; Hodgson et al. 2007; Northridge et al. 2011). As pingers might deter some cetaceans while attracting some pinnipeds, addressing a bycatch issue is likely to be challenging if more than one species is at risk and they have conflicting responses to pingers (Mackay and Knuckey 2013).

For pot/trap fisheries (Appendix 2E), pinger effectiveness in deterring large whales from high-risk entanglement areas, particularly pot or trap fishery operations, appears to be variable depending on species, migration direction and social category. In Canadian inshore trap fisheries, acoustic devices appeared to reduce the collision frequency between humpback whales (*Megaptera novaeangliae*) and cod traps (Lien et al. 1992). However, in Australia, while southward migrating humpback whales exhibited aversion behaviour to acoustic stimuli (Dunlop et al. 2013), northward migrating whales showed no detectable response to pingers (Harcourt et al. 2014; Pirotta et al. 2016).

There were indications that pingers could potentially deter grey whales (*Eschrichtius robustus*) from high risk coastal areas, although results were inconclusive due to low statistical power (Lagerquist et al. 2012).

Ensuring pingers are functioning correctly and with the required number in the correct net location is important for maintaining effectiveness in gillnet fisheries (Orphanides and Palka 2013). However, the financial cost of implementing pingers may limit their applicability in many developing countries and/or smaller fisheries (Dawson et al. 2013; Dawson et al. 1998; Read 2008), and more cost-effective, durable pingers are needed (Crosby et al. 2013). Pingers are also unlikely to be effective in deterring dolphins if they are not fully functional (e.g. fully charged batteries) or in suboptimal locations on trawl gear (Deepwater Group 2018; Northridge et al. 2011), and they should be positioned to ensure they do not impact operational equipment, such as net monitoring systems (Morizur et al. 2007).

Evidence of harbour porpoise habituation to pingers, which would reduce their effectiveness in mitigating gillnet bycatch, was provided by some experimental studies (Carlstrom et al. 2009; Cox et al. 2001; Dawson et al. 2013; Gearin et al. 2000; Read 2013), but not others (Hardy et al. 2012). However, long-term studies monitoring operational gillnets showed no sign of harbour porpoise, common dolphin or beaked whale habituation to pingers (Carretta and Barlow 2011; Dawson et al. 2013; Palka et al. 2008). Inshore, resident porpoise populations may be more likely to develop habituation to pingers than more migratory species (Dawson et al. 2013; Dawson et al. 1998). The effectiveness of pingers in deterring coastal, inshore or river finless porpoises (*Neophocaena* spp.) from gillnets decreased after a few months, and developing regimes which include periods with no pinger use (Amano et al. 2017), as well as randomising pinger frequency, time interval and strength, may help to maintain effectiveness. Developing 'responsive pingers' for gillnets, which only emit sounds in response to cetacean echolocations, may reduce the likelihood of pinger habituation for some species (Leeney et al. 2007; Waples et al. 2013). Bottlenose dolphins may become *more* sensitised to pingers, which could increase the mitigation effect on this species over time (Cox et al. 2003). With respect to trawl gear, some captive pinniped species became habituated to pingers on a simulated net and continued to depredate netted fish, while some dolphin species charged the netting despite pinger presence (Bowles and Anderson 2012). An interactive pinger for pelagic trawls, designed to emit signals in response to the presence of dolphin echolocations, may delay habituation and reduce noise pollution in the marine environment, with initial tests showing evasive behavioural responses from bottlenose dolphins, although not from common dolphins (van Marlen 2007). In longline operations, there is evidence that false killer whales (*Pseudorca crassidens*) and killer whales (*Orcinus orca*) became habituated to acoustic devices (Mooney et al. 2009; Tixier et al. 2015).

The increasing level of anthropogenic sound in the marine environment may negatively impact the behaviour, physiology and auditory systems of some marine species (Kastelein et al. 2015), with indications that some gillnet pingers may affect target and non-target fish (Goetz et al. 2015; Kastelein et al. 2007). Pinger deployment could impact small cetacean species that are neophobic and with small, restricted ranges by excluding them from crucial habitat, with the displacement effect potentially more pronounced in coastal locations where topographical features limit access to key bodies of water (Dawson et al. 2013). In longline operations, there is concern that frequent exposure to higher amplitude devices may affect the echolocation ability of killer whales (Tixier et al. 2015).

Weakened gear

Different types of weakened gear, designed to release caught animals, have been proposed and/or trialled in different fisheries (Table 2.1) including:

a) *"Weak" hooks* in longline fisheries (Appendix 2C): These may reduce the bycatch risk for some species (e.g. false killer whales) without loss of target catch (Bayse and Kerstetter 2010; Bigelow et al. 2012; McLellan et al. 2015; Werner et al. 2015), although there is currently insufficient evidence to support this. Low rates of cetacean interactions during experimental trials has hampered the ability to assess bycatch reduction (Bigelow et al. 2012). Weak hooks would not reduce interactions or prevent depredation (Hamer et al. 2015; Werner et al. 2015).

b) *Reduced-strength nets or ropes*: Thin twine gillnets may significantly reduce seal and harbour porpoise bycatch compared to thick twine nets (Northridge et al., 2003) (Appendix 2D). Similarly, as strong polypropylene ropes used in modern pot/trap fisheries have increased the mortality risk of entangled cetaceans, use of ropes with reduced breaking strengths could substantially decrease mortalities of whales entangled in fixed gear (Knowlton et al. 2016) (Appendix 2E).

c) *Weak links* between the vertical line from a pot/trap to a buoy: These do not appear to have reduced the incidence or severity of whale entanglements in USA lobster fisheries (Knowlton et al. 2012; Knowlton et al. 2016; Pace et al. 2014; Salvador et al. 2008; Van der Hoop et al. 2013) (Appendix 2E). Also, when buoys separate from vertical pot or trap lines, released whales may retain sections of gear (Laverick et al. 2017; Moore 2009). Some USA fisheries require weak links in gillnets to allow entangled whales to break free (NOAA 2018), although no research was identified that tested the efficacy of this measure (Appendix 2D).

2.4.2 Trawl

Marine mammals are frequently caught in pelagic or midwater trawls as these often target the same pelagic species eaten by marine mammals, have relatively high tow speeds with large nets, and usually operate within marine mammal diving ranges for extended periods (Fertl and Leatherwood 1997; Hall et al. 2000) (Appendix 2A). However, in US fisheries, marine mammals are caught more often in demersal rather than midwater trawls (Carretta et al. 2017; Jannot et al. 2011; Waring et al. 2016). The technical mitigation measures identified and assessed for trawls, in addition to pingers (see section 1.1), are *net colour, net binding*, *exclusion devices*, *rope or mesh barriers* and *auto-trawl systems* (Table 2.1; Appendix 2A).

Net colour

In an Australian fishery, more bottlenose dolphins were caught in a grey trawl net compared to a standard green net, although management variations between the two trial vessels, resulting in different net speeds through the water during winching, could also have contributed to bycatch differences (Stephenson and Wells 2006). Changing net colour has not been tested as a means of reducing marine mammal bycatch risk. However, this may not be a feasible mitigation option as, particularly for some small cetacean and fur seal species that are known to deliberately enter nets to depredate the captured fish (Fertl and Leatherwood 1997; Hamer and Goldsworthy 2006; Lyle et al. 2016; Wakefield et al. 2017), bycatch risk may not be linked to their lack of awareness of a trawl net's presence. Visual detection of nets may also be limited if visibility is poor or variable at fishing depths. Furthermore, as well as vision, many cetacean species may primarily rely on echolocation to forage and pinnipeds may use tactile senses (Martin and Crawford 2015).

Net binding

An organic material, such as sisal string, is used to bind the net until it has sunk below the water surface. Once the trawl doors are paid away, the water force separating the doors breaks the bindings so the net can form its standard operational position. Net binding, used to mitigate seabird bycatch during net shots (Sullivan et al. 2004), has also been used in some Australian fisheries to reduce fur seal (*Arctocephalus* spp.) interactions during setting (Australian Fisheries Management Authority, personal communication), although there is a lack of operational information or testing to determine whether this effectively reduces seal bycatch. As marine mammal interactions often occur during the haul (Hamer and Goldsworthy 2006), net binding, if it is shown to be effective, may need to be used in combination with other mitigation.

Exclusion devices with separation grids

It is widely accepted that appropriately designed exclusion devices successfully prevent mortalities of a range of non-target marine species in nets without significantly impacting target catch (Dotson et al. 2010; Griffiths et al. 2006; Hamilton and Baker 2015a; Wakefield et al. 2017; Zeeberg et al. 2006), although there are differing outcomes for pinnipeds and cetaceans. The grid design and escape hole configuration of exclusion devices need to ensure target species flow smoothly into the codend without compromising catch quality and quantity (Table 2.2), while ensuring all size classes of the non-target marine mammal species are prevented from passing into the codend and can escape (Hamilton and Baker 2015a). In fisheries with large target species, designing grids that have no impact on target catch is likely to be more challenging.

Top-opening, hard-grid exclusion devices (Figure 2A.1 in Appendices) have effectively reduced pinniped bycatch in a number of trawl fisheries (CCAMLR 2017; Hamilton and Baker 2015a; Lyle et al. 2016; Tilzey et al. 2006). Operational constraints may influence exclusion device design, which could limit bycatch reduction. For example, on-board net drum storage may necessitate top-opening devices to have flexible grids, as an upwardly angled grid is counter to net drum rotation. However, soft-grids deformed under a seal’s weight causing partial entanglements, provided no passive assistance in directing seals out an opening, and flexible grid distortion may also restrict the flow of target species into the codend resulting in reduced catches (Bord Iascaigh Mhara and University of St Andrews 2010; Lyle et al. 2016) (Table 2.2).

There has been limited success in demonstrating exclusion devices effectively reduce cetacean bycatch. Dolphins may deliberately enter trawl nets to depredate captured fish but do not appear to manoeuvre as easily as pinnipeds within the confines of a net (Jaiteh et al. 2013; Lyle et al. 2016). They appear to become distressed when far into nets and unable to find, or negotiate escapes, particularly those with bottom-opening exits (Jaiteh et al. 2014; Wakefield et al. 2017; Zeeberg et al. 2006). While there are reports of bottlenose dolphins seeming to favour an exit out the bottom of a net (Zollett and Rosenberg 2005), they have also been reported to preferentially attempt escape via the net mouth rather than exclusion devices and, therefore, may be more likely to die if progressing too far into a net (Wakefield et al. 2017). Exclusion devices showed potential in reducing common dolphin bycatch in UK midwater pair trawls (Northridge et al. 2011), although only a small number successfully exited via the escape hole with most appearing to detect the grid some distance beforehand and attempting to, unsuccessfully, escape in that area (Bord Iascaigh Mhara and University of St Andrews 2010). While ensuring no impact on target catch, net drag or operational functioning, it was thought that positioning exclusion devices as far forward as practical, with multiple, obvious escape routes, may be critical for small cetacean survival (van Marlen 2007).

Unobservable and unreported *cryptic* mortality may occur with exclusion devices due to injuries incurred during interactions with devices or because dead animals may fall out escape openings, although scientific evidence has shown that cryptic mortalities from direct interactions with top-opening, hard-grid exclusion devices are unlikely (Hamilton and Baker 2015a; Hamilton and Baker 2015b). A forward-facing hood, held in place with a 'kite' (i.e. material strip) and floats over a top-opening escape (Figure 2A.1 in Appendices), directs water flow into the net across the grid and is likely to minimise potential loss of dead or incapacitated animals and target catch, while keeping the escape hole open and assisting live animals to escape (Hamilton and Baker 2015a). Target species, dead seals and dead dolphins have been observed falling out of devices with *bottom-opening* escapes, or top-opening escapes *without a cover or hood* (Hamilton and Baker 2015a; Jaiteh et al. 2014; Lyle et al. 2016; Stephenson and Wells 2006), although unaccounted mortality was considered negligible even with bottom-opening devices (Wakefield et al. 2014; Wakefield et al. 2017). While Lyle et al. (2016) stated that passive ejection of dead animals had been reported for top-opening devices citing Robertson (2015) and Wakefield et al. (2014), there is no evidence to support this. Robertson (2015) stated there were no data to show dead sea lions were either retained or passively ejected from openings, but made no differentiation between top-opening and bottom-opening devices and did not acknowledge that a hood or cover helps prevent passive loss of animals (see Hamilton and Baker 2015b). Wakefield et al. (2014) reported one incident where a dead dolphin fell out a device with a top-opening escape hole, although this occurred when the net rotated 180° during the haul so the hole (with no cover) was orientated downward.

Rope or mesh barriers

Restricting dolphin access into trawl nets may be the key to preventing mortality, although there has been limited success in deterring them from entering nets (Wakefield et al. 2017). A small number of dolphins escaped through a top-opening hole, covered with parallel 'bungee' cords, located ahead of a mesh barrier, though most barriers trialled in pair trawls (e.g. various designs in van Marlen, 2007) had reduced target catch rates (Bord Iascaigh Mhara and University of St Andrews 2010; van Marlen 2007) (Table 2.2).

Auto-trawl systems

Intuitively, ensuring the net entrance does not collapse during any trawl phase should reduce entrapment risk and maintain the effective operation of installed exclusion devices, though the efficacy of auto-trawl systems as a bycatch mitigation measure requires verification. Use of otter-board sensors and eliminating sharp turns while trawling are thought to have reduced dolphin mortalities in trawl nets (Wakefield et al. 2017). Recent research assessing bottlenose dolphin interactions with trawls gives further support to improving and monitoring trawl gear stability as potentially the most effective mitigation strategy for reducing dolphin bycatch, while the use of acoustic deterrent devices was ineffective (Santana-Garcon et al. 2018).

2.4.3 Purse seine

In purse seine operations, bycatch mitigation has concentrated on reducing dolphin mortality, mainly related to eliminating the practice of setting around dolphin pods associated with target tuna species in the eastern Pacific Ocean (EPO) (Gilman 2011) (Appendix 2B). Less commonly, sets on tuna schools associated with live whales have also occurred. Reducing cetacean bycatch has been mainly through the cessation of sets on dolphin-associated or whale-associated tuna schools (Hall and Roman 2013). There was a general lack of information on mitigation development for other purse seine fisheries although a '*dolphin gate*' (detachable cork-line section) and weights to help sink the cork-line were trialled in an Australian small pelagic fishery but require further development and testing to determine if effective (Hamer et al. 2008). In the EPO tuna fisheries, a shift to sets around 'Fish Aggregating Devices (FADs)' (i.e. artificial floating elements with relocation aids) raised new environmental concerns regarding overfishing and marine species' entanglement in FAD components (Hall and Roman 2013). Mitigation development currently focuses on improving FAD design to reduce shark and turtle entanglement (Restrepo and Dagorn 2011; Restrepo et al. 2014; Restrepo et al. 2016), while this appears less of an issue for marine mammals. In terms of technical mitigation, to reduce bycatch and increase the likelihood of dolphin escape, the use of *enlarged mesh sizes* was unsuccessful as target species and dolphins are often similar size, and *acoustic methods* to frighten dolphins out of nets (e.g. playback of alarms calls or killer whale sounds) were also ineffective (Gabriel et al. 2005). The primary mitigation that has substantially reduced dolphin mortality in tuna purse seine fisheries, without causing loss of entrapped tuna (Restrepo et al.

2016), has been the *'back-down' manoeuvre* with the addition of *'Medina' panels* as described by Hall and Roman (2013) (Table 2.1). Speedboats equipped with towing bridles can also help keep the net open and assist dolphin escape as well as the use of a raft inside the net to facilitate manual rescue (National Research Council 1992). While 'cryptic' impacts, including the post-escape mortality of injured dolphins and potential effects of chasing and encirclement on reproductive success, are a potential issue (Anderson 2014; Archer et al. 2004; Cramer et al. 2008; Gerrodette and Forcada 2005; Wade et al. 2007), no studies were identified that monitored the post-release survival of marine mammals.

2.4.4 Longline

While mitigation has primarily focussed on reducing the economic impact of marine mammal depredation of target catch in longline operations, depredation behaviour also puts them at risk of becoming hooked or entangled (Bigelow et al. 2012) (Appendix 2C). Mitigation measures that have been unsuccessful in reducing interactions include the use of *explosives, chemical deterrents, flare guns* or *predator sounds* (Werner et al. 2006). An assessment of mitigation measures including spatial fisheries management, altered fishing strategies, and acoustic and physical techniques, concluded that terminal gear modifications had the greatest mitigation potential (Werner et al. 2015). In this updated review, along with pingers and weak hooks (see sections 1.1 and 1.2), mitigation with potential to reduce interactions with longlines are *passive acoustic deterrents* and *catch protection devices* (Table 2.1; Appendix 2C).

Passive acoustic deterrents

Echolocation Disruption Devices and passive acoustic measures may affect a cetacean's ability to echolocate hooked fish (Hamer et al. 2012; O'Connell et al. 2015). While there were indications that spherical beads attached near longline hooks could reduce interactions between sperm whales (*Physeter microcephalus*) and longlines, it was inconclusive whether they were effective (O'Connell et al. 2015).

Catch protection devices – demersal longline

In demersal longline fisheries, odontocetes are more likely to access hooked fish during the haul compared to line soaking that may be at depths beyond their normal foraging range (Gilman et al. 2006; Guinet et al. 2015; Hamer et al. 2012; Soffker et al. 2015; Tixier et al. 2014). However, there may be exceptions to this such as recent evidence of interactions between southern elephant seals (*Mirounga leonina*) and a sub-Antarctic demersal longline fishery during the line soak period at depths >1 km (van den Hoff et al. 2017). "Net sleeves", which cover hooked fish with the downward pressure of hauling, protect fish from depredation and reduce bycatch risk (Hamer et al. 2012; Moreno et al. 2008). The "cachalotera", a type of net sleeve, substantially reduced depredation by killer and sperm whales in the Chilean industrial Patagonian toothfish (*Dissostichus eleginoides*) longline fleet (Moreno et al. 2008) (Figure 2C.1 in Appendices), although some killer whales have learnt how to depredate around cachaloteras (Arangio 2012). A similar system to reduce sperm whale depredation and seabird bycatch on Spanish vessels consists of "umbrella" devices fixed on branchlines, which open to extend over hooked fish, combined with stones for faster line sinking (Figure 2C.2 in Appendices). While "umbrella and stones" net sleeve trials were promising, evidence for their efficacy in reducing interactions was inconclusive (Goetz et al. 2011). While there was no reduction in target catch rates with "cachaloteras" (Moreno et al. 2008), "umbrella and stones" net sleeves significantly reduced toothfish catch, which may be due to different attachment designs as "cachaloteras" slide up and down the branchline whereas the "umbrellas" are fixed (Goetz et al. 2011). In some operations, gear and vessel configurations (e.g. if hooks are close together and gear is coiled for storage) may make net sleeves impractical to use (O'Connell et al. 2015).

Triggered catch protection devices – pelagic longline

Compared to demersal longlines, pelagic longlines may be at risk of depredation during setting, soak time and hauling as marine mammals often occur across the same depths as target fish and, therefore, net sleeves that slide over the hook only during the haul have limited use (Hamer et al. 2015; Rabearisoa et al. 2015). Therefore, devices developed for pelagic longlines have mechanical triggers to release the net sleeve structure with the pressure when a fish is hooked. These include:

a) "Chain" devices and cone-like "cage" devices (Figure 2C.1): Trials on Australian pelagic longlines showed all odontocete interactions occurred on branchlines without devices and there was negligible impact on target catch, although results were inconclusive due to small sample sizes (Hamer et al. 2015);
b) Eight strand "spider" devices and conical "sock" devices (see photographs in Rabearisoa et al. (2012)): Trials on commercial tuna longliners off the Seychelles showed limited success in reducing odontocete depredation, although interaction rates were low during trials (Rabearisoa et al. 2012);
c) "DEPRED" device (Figure 2C.4): Initial results were encouraging although, as trials used small delphinid interactions with a small pelagic fish fishery as a proxy

for odontocete interactions with tuna and billfish fisheries, further development and testing is required (Rabearisoa et al. 2015).

During trials, some devices falsely triggered when a fish was not present, did not deploy when a fish was hooked, or became entangled in the longline gear (Hamer et al. 2015; Rabearisoa et al. 2012). While most devices provide a simple physical barrier to protect hooked fish, there has been preliminary testing of devices with metal wire incorporated in streamers to affect an odontocete's ability to echolocate hooked fish (McPherson et al. 2008).

2.4.5 Gillnet

There have been a number of reviews, with a range of focuses and objectives, relating to marine mammal bycatch and mitigation measures for gillnets (Dawson et al. 2013; Leaper and Calderan 2018; Mackay and Knuckey 2013; Northridge et al. 2017; Read 2008; Read 2013; Reeves et al. 2013; Uhlmann and Broadhurst 2015; Waugh et al. 2011). As it may be difficult for many marine mammal species to avoid gillnets, well-designed spatial and/or temporal fishery closures are likely to be important and effective mitigation options (Dawson and Slooten 2005; Hall and Mainprize 2005; Hamer et al. 2013; Hamer et al. 2011; Read 2013; Rojas-Bracho and Reeves 2013; Slooten 2013). In this updated review, in addition to pingers, reduced strength nets and weak links (see sections 1.1 and 1.2), the potential mitigation options identified are *acoustically reflective nets*, *visually detectable nets* and *"buoyless" nets* (Table 2.1; Appendix 2D).

Acoustically reflective nets

As nylon may be difficult for echolocating marine mammals to detect, nets that utilise different materials, or incorporate reflective components (e.g. metal compounds) into the net filament, have been trialled (Bordino et al. 2013; Larsen et al. 2007; Mooney et al. 2004; Trippel et al. 2003). Some studies showed a reduction in harbour porpoise bycatch with metal oxide nets (Larsen et al. 2007; Trippel et al. 2003; Trippel et al. 2008) while others reported no reduction in harbour porpoise, franciscana or seal bycatch (Bordino et al. 2013; Mooney et al. 2004; Northridge et al. 2003). It was suggested that observed bycatch reduction may be due to net stiffness rather than acoustic reflectivity (Cox and Read 2004; Larsen et al. 2007). However, while increasing net stiffness could be a low-cost mitigation option (Northridge et al. 2017), there was no significant difference in franciscana bycatch between barium sulphate nets, nets with increased nylon twine stiffness and standard nets (Bordino et al. 2013).

Furthermore, increased net stiffness decreased target catch rates in some studies (Larsen et al. 2007). Increasing a net's acoustic reflectivity would also be ineffective if a small cetacean encountered the net when it was not echolocating (Dawson 1991). There was also no evidence that passive acoustic additions (e.g. metal beaded chains) reduced cetacean bycatch (Hembree and Harwood 1987).

Visually detectable nets

Increasing the visual detectability of nets using illumination or visible panel inserts have not yet been tested as a mitigation option for marine mammals. Light-emitting diodes significantly reduced bycatch of other taxa and could potentially reduce the bycatch of small cetaceans (Mangel et al. 2018). Conversely, installing contrasting patterned panels to increase net detectability could possibly alert pinnipeds to gillnet presence which may increase catch depredation (Martin and Crawford 2015). To date, changing gillnet colour has not been tested as a measure to reduce marine mammal bycatch, although orange-coloured gillnets may be more apparent to some penguin species (Hanamseth et al. 2018).

"Buoyless" nets

Nets with reduced numbers of buoys per metre significantly reduced sea turtle bycatch probably due to a decreased vertical profile of the nets. While this gear modification could potentially reduce marine mammal bycatch (Peckham et al. 2016), this is yet to be verified.

2.4.6 Pot/Trap

Management of large whale entanglement has predominantly focussed on strategies to respond and release entangled whales or establish seasonal closures (Robbins et al. 2015; Van der Hoop et al. 2013), with less research on technical solutions to prevent interactions or entanglements (Appendix 2E). Developing better species-specific knowledge of the interaction and mechanism of entanglement, particularly the parts of gear that whales mainly encounter, will aid in implementing effective mitigation (Johnson et al. 2005; Northridge et al. 2010). A number of potential techniques have been proposed but have not been considered a priority for development including ropes that glow underwater and lipid soluble ropes which dissolve if embedded in whale blubber (Werner et al. 2006). This review provides an updated assessment of entanglement mitigation based on previous reviews (Laverick et al. 2017; Leaper and Calderan 2018), as well as mitigation to reduce pinniped and small cetacean entrapment in pots and traps. In addition to pingers, reduced strength rope, weak links and line cutters (see sections 1.1 and 1.2), the mitigation identified are *pot/trap* *excluder devices*, *“seal socks”*, *sinking groundlines*, *rope-less pot/trap systems*, *rope colour changes* and *stiff ropes* (Table 2.1; Appendix 2E).

Pot/trap excluder devices

Technical alterations or additions reduce the entrance size and/or shape of pots and traps to prevent marine mammals entering thereby reducing bycatch risk as well as catch depredation. The shape as well as the size of pot entrances is likely to be important to ensure target catch quantity and size range are not affected (Konigson et al. 2015). ‘Bungee’ cord guards reduced bottlenose dolphin interactions with crab pots (Noke and Odell 2002; Werner et al. 2006); wire guards and stronger netting reduced seal damage and bycatch risk in salmon trap-nets (Hemmingsson et al. 2008; Suuronen et al. 2006); and smaller crab fyke trap openings reduced sea otter (*Enhydra lutris*) bycatch without reducing target catch (Hatfield et al. 2011). ‘Spike’ excluder devices are mandatory in bycatch risk areas to prevent Australian sea lion pups and juveniles entering lobster pots, although testing of alternative industry-designed 150 mm diameter circular openings (which are more practical and safe to use) has shown they may also effectively exclude most pups (Campbell et al. 2008; Mackay and Goldsworthy 2017).

“Seal socks”

A cylindrical net attached to shallow water (< 2 m deep) fyke nets allowed trapped seals access to the surface to breathe and reduced ringed seal (*Phoca hispida botnica*) bycatch, although was less effective for Baltic grey seals (*Halichoerus grypus baltica*) (Oksanen et al. 2015).

Sinking groundlines

The implementation of measures in USA fixed-gear fisheries, including negatively buoyant or sinking groundlines which aim to lie closer to the ocean bottom, has not reduced serious injuries and mortality of northern right whales (*Eubalaena glacialis*) to sustainable levels (Brillant and Trippel 2010; Knowlton et al. 2012).

Rope-less pot/trap systems

To reduce cetacean entanglement risk, rope-less systems remotely release buoys linked to pots or traps, thereby reducing surface markers with vertical lines in the water column. There are no published studies that show rope-less systems mitigate bycatch or are practical and cost-effective for implementation in operational fisheries (Laverick et al. 2017), although trials have been undertaken on rope-less system prototypes using timed-release (Partan and Ball 2016) or acoustic-release mechanisms (How et al. 2015; Salvador et al. 2006; Turner et al. 1999). Acoustic releases have been used for some years in an Australian lobster fishery (Liggins 2016), although research on acoustic release technology in this fishery (Hodge 2015) is yet to be published. Acoustic-release systems may be preferable to pre-specified time-release mechanisms which may release ropes before fishers are in the vicinity to haul gear (How et al. 2015; Laverick et al. 2017).

Rope colour changes

Preliminary studies showed northern right whales visually detected red and orange ‘simulated’ ropes at greater distances than black and green ropes (Kraus et al. 2014; Kraus and Hagbloom 2016), which suggested that changing to red and/or orange commercial fishing ropes may improve their ability to avoid entanglements. However, an over-representation of yellow and orange ropes in humpback whale entanglements in Australia may indicate this species actively target these ropes or, in contrast to right whales, yellow and orange are less visually detectable to humpback whales (How et al. 2015). Minke whales appeared to detect black and white ropes more easily than other colours (Kot et al. 2012).

Stiff ropes

Although increasing rope stiffness could reduce entanglement risk as whales may be able to glide off stiff ropes more easily, there are no published studies on whether ropes with greater stiffness or tension reduce entanglements (Consortium for Wildlife Bycatch Reduction 2014). However, experimental testing using a model of a right whale flipper indicated that stiff ropes may increase injuries at the point of contact (Baldwin et al. 2012).

2.5 Conclusions and recommendations

Trawl – conclusions and research needs

Fishery-specific variables and issues need to be considered when designing exclusion devices including the size, biology and behaviour of non-target and target species; size, operation and storage of gear; towing speed; and trawl hydrodynamics in relation to net size/grid and escape hole ratios. Exclusion device grid construction (material, grid angle, bar spacing and size); escape hole size, shape and location (e.g. top or bottom); and the addition of a cover or hood, are all important components that will impact bycatch reduction efficacy (Baker et al. 2014). Appropriately designed exclusion devices have effectively reduced pinniped bycatch in trawl nets. In particular, devices with hard separation grids angled to top-opening escape holes, with a cover or hood held open by a kite and floats, effectively allow pinnipeds to escape and post-escape mortality is likely to be low (Hamilton and Baker 2015a). Loss of dead animals out top-opening holes with covers is considered unlikely, although this requires further verification, ideally by direct assessment of pinniped interactions with exclusion devices in operational fisheries. While there has been limited success with bottom-opening devices, air-breathing marine mammals are probably less likely to escape downwards (Allen et al. 2014) and, furthermore, bottom-opening devices, particularly without covers, may be more likely to have unreported bycatch from dead animals dropping out.

Exclusion devices are not fully effective in mitigating cetacean bycatch in trawl fisheries. Research is required on options for reducing cetacean bycatch including further information on the escape behaviour of dolphin species that interact with nets to inform the optimal location for exclusion devices (probably further forward in nets) and ensure escape options are clear, while retaining target catch (van Marlen 2007). It is inconclusive whether rope or mesh barriers prevent entry of small cetaceans past the fore section of trawls, thereby, reducing bycatch. Furthermore, barriers may reduce target catch to unacceptable levels (Bord Iascaigh Mhara and University of St Andrews 2010).

Net binding may be effective in reducing bycatch risk during the shot, although would only be feasible in operations where the net is removed from the water and brought onto the trawl deck after each trawl. The efficacy of net binding in reducing marine mammal bycatch requires testing, including research to establish the optimal technical specifications to ensure the net remains bound until it reaches depths beyond the diving range of bycatch species. Net binding would only potentially reduce interactions during net shooting and is likely to be ineffective for mitigating bycatch of deep-diving species.

Loud pingers show promise in reducing small cetacean interactions with trawl gear, particularly for common dolphins (Northridge et al. 2011), although may not be effective for bottlenose dolphins (Santana-Garcon et al. 2018). However, development of more robust and operationally manageable devices is required as well as more fishery-specific testing to determine the optimal configuration and spacing of pingers in trawl operations and verification that pingers significantly deter dolphins (Bord Iascaigh Mhara and University of St Andrews 2010; Northridge et al. 2011; van Marlen 2007). Investigating the likelihood of cetacean habituation to pingers as well as the impact of the widespread use of loud pingers on the behaviour, distribution and ecology of cetaceans and other marine species is also needed (Northridge et al. 2011).

Maintaining the shape and structure of trawl nets may be an integral bycatch mitigation strategy, particularly for cetaceans (Santana-Garcon et al. 2018). Auto-trawl systems potentially mitigate bycatch by ensuring the net entrance is always open thereby reducing entrapment risk, although this needs investigation and validation. However, as these systems are routinely used by some trawlers to improve fishing efficiency, evaluation of their mitigation potential in an experimental framework may be difficult.

Purse seine – conclusions and research needs

Management measures, particularly the ‘back-down’ manoeuvre coupled with ‘Medina’ safety panels and additional guidance from small boats, increase the safe escape and have significantly reduced the observed bycatch of small cetaceans in tuna purse seine fisheries. However, information is needed on the post-encirclement and post-release survival and health of bycatch species through remote monitoring programs to inform best practice techniques for releasing encircled animals (Restrepo et al. 2014). Although potentially less relevant to marine mammal species, continued research on the development and efficacy of non-entangling FADs is also important.

Longline – conclusions and research needs

There is a lack of technical mitigation shown to be fully effective in reducing marine mammal bycatch in longline fisheries. However, there are indications that catch protection devices, with specific designs for both pelagic and demersal operations, reduce hooking risk for odontocetes. Results have been variable on the impact of different net sleeve devices on target catch rates, and more research is required, particularly to reduce interactions with killer whales that have learnt to get around standard designs in demersal longline operations (Arangio 2012; Goetz et al. 2011; Moreno et al. 2008). In pelagic longline operations, further research is required to refine triggered catch protection device designs, particularly increasing device reliability, and verifying mitigation efficacy in operational fisheries in the longer term (Hamer et al. 2015; Rabearisoa et al. 2015; Rabearisoa et al. 2012).

While the use of weak hooks may reduce bycatch in pelagic longlines, this requires further operational testing, including operational feasibility. There is also a lack of information on the post-release health and survival of marine mammals that are injured, retain or ingest hooks, or remain entangled in gear (Bayse and Kerstetter 2010;

Hamer et al. 2012; Hamer et al. 2015; Hucke-Gaete et al. 2004; Kock et al. 2006; Werner et al. 2015).

Gillnet – conclusions and research needs

Pingers effectively reduce the gillnet bycatch of some (e.g. harbour porpoises), although not all, small cetacean species, and may be most effective in reducing bycatch of neophobic species with large home ranges (Dawson et al. 2013). Pinger research should include evidence that target species size and catch are not impacted (Barlow and Cameron 2003; Carlstrom et al. 2002; Gearin et al. 2000; Kraus et al. 1997; Larsen and Eigaard 2014; Waples et al. 2013). While the evidence is that harbour porpoises do not become habituated to pingers (Dawson et al. 2013; Palka et al. 2008), further investigations regarding habituation for other cetacean species are needed. More research to understand small cetacean behaviour in response to 'reactive pingers' is also required, particularly if they may reduce the likelihood of habituation and potential impacts from marine noise pollution (Leeney et al. 2007). As pingers rely on changing animal behaviour to avoid nets, they should only be implemented after rigorous fishery-specific research on the impacts on all likely bycatch species (Hodgson et al. 2007) and other vulnerable species within the ecosystem. The long-term effects of pinger exposure on small cetaceans, particularly exclusion from key habitat areas, is not well known. Care should be taken when deploying pingers to mitigate bycatch in areas with ecologically important small cetacean habitat, and intensive pinger use in coastal areas should be carefully monitored (Carlstrom et al. 2002; Carlstrom et al. 2009; Kyhn et al. 2015). Operational testing should include research on the optimal positioning and spacing of pingers and, following implementation, ongoing monitoring is required to maintain pinger effectiveness. As commercially available pingers may be prohibitively expensive in some fisheries, more cost-effective solutions are required. The development of more durable pingers with battery change capabilities may help to reduce implementation costs (Crosby et al, 2013).

There have been conflicting results on the effectiveness of acoustically reflective metal oxide nets in reducing small cetacean bycatch, and further research is needed to better understand the mechanism of why some metal oxide nets showed bycatch reduction (Northridge et al. 2017).

Increased research focus is needed on post-release impacts following direct interactions with gillnets. For example, pinnipeds and cetaceans released following entrapments in deep-set gillnets (and trawls) may incur gas embolism that could lead to post-release mortality (Fahlman et al. 2017; Moore et al. 2009).

Pot/Trap – conclusions and research needs

Fishery-specific trap guards or 'excluder devices' have been effective in reducing the entrapment risk of marine mammals while maintaining target catch rates (Campbell et al. 2008; Konigson et al. 2015; Noke and Odell 2002). The use of "seal socks" may be a potential mitigation option in shallow-water fyke net fisheries, although may not be effective for all pinniped species (Oksanen et al. 2015).

Results on the effectiveness of pingers in deterring large baleen whales from potentially high-risk areas have been variable (Dunlop et al. 2013; Lien et al. 1992; Pirotta et al. 2016) and species-specific investigations of different pingers are required to determine if some designs may be more consistently effective. However, identifying a lack spatial *deterrent* behaviour relative to a pinger in experimental trials may not necessarily mean that pingers would be ineffective in *alerting* marine mammals and reducing operational interactions (McPherson 2017).

'Rope-less' buoy systems are a promising mitigation development, although further design refinement and efficacy research is required. Acoustic-release systems may be preferable to timed-release systems but are likely to have higher establishment costs, and research is needed on reliable deployment systems and a device with enough rope for fisheries operating in deep water (How et al. 2015; Partan and Ball 2016; Salvador et al. 2006).

While ropes with reduced breaking strength could substantially decrease whale mortality in fixed gear, research is required on the practicalities and success of using reduced-strength ropes in operational fisheries (Knowlton et al. 2016), and the post-release health and survival of animals that remain entangled in lines or sections of gear (Werner et al. 2015). The effectiveness of weak links in buoy lines needs investigation due to concerns regarding a lack of reduction in whale entanglements following weak link implementation in USA lobster fisheries (Knowlton et al. 2012; Pace et al. 2014; Van der Hoop et al. 2013). It is noteworthy that weak links are not recommended in some Australian fisheries as disentangling 'anchored' whales from gear has been more successful than locating and disentangling free-swimming whales (How et al. 2015).

Whale responses to different rope colours appears to be species-specific (How et al. 2015; Kot et al. 2012; Kraus et al. 2014; Kraus and Hagbloom 2016). Further species-and fisheries-specific research is needed to test and understand whale detectability of colours in a range of conditions (Kraus et al. 2014).

Final summary and conclusions

Effective technical mitigation measures are a crucial element of any robust, integrated bycatch management program, which usually includes other management directives such as temporal and spatial fishing restrictions and appropriate operational 'codes of practice'. For some gear types and taxa, there are currently limited technical options with strong evidence they effectively reduce bycatch, and substantial development and research of best practice mitigation options is needed to address marine mammal bycatch in many fisheries. For mitigation to be considered effective, a significant reduction in bycatch mortality needs to be demonstrated, together with maintenance of target catch quality and quantity. Fishing industry engagement to ensure design, development and effective implementation of practical solutions is also essential. Therefore, knowledge of the biological and behavioural characteristics of target and bycatch species, temporal and spatial overlap of bycatch species with fishing activities and operational factors is needed (Baker et al. 2014). Determining mitigation efficacy should include species- and fisheries-specific testing with adequate scientific rigour, and a quantitative target to enable efficacy assessment.

The reviewed studies varied greatly in the level and rigour of scientific testing to verify mitigation effectiveness in reducing bycatch. Some measures have undergone controlled studies in a range of conditions (e.g. pingers to reduce harbour porpoise bycatch, Dawson et al. (2013)), while others have not been tested, testing has been inadequate, or experimental design has been inappropriate (e.g. Dawson and Lusseau (2005)). However, testing can be difficult as a technical measure may be implemented as part of a suite of management actions, confounding attempts to test its specific effectiveness in reducing bycatch (Laverick et al. 2017). Ideally, if efficacy is to be efficiently demonstrated, mitigation needs to be tested against a control of no-deterrent, although such trials are often difficult to implement for ethical reasons. Additionally, the logistics of undertaking controlled studies in operational fisheries, including low or sporadic marine mammal interaction rates during trials, may limit the scientific robustness of testing (Dawson et al. 1998; Hamer and Goldsworthy 2006). Obtaining adequate data from comparable controlled experiments may be particularly challenging in trawl fisheries with small numbers of vessels towing a single net. Furthermore, due to the range of variables during fishing (e.g. location, weather, season, ecosystem components), controlled experiments of the same mitigation for the same bycatch species may produce conflicting results in different operations. Technical measures experimentally shown to be effective also require post-implementation monitoring in operational fisheries, and mitigation may not produce the same bycatch decrease in an operational fishery as shown in controlled trials (Orphanides and Palka 2013). Ensuring the ongoing effectiveness of implemented mitigation requires fisheries to maintain adequate observer coverage (either direct observations, or electronic monitoring and review), continue correct deployment of the appropriate measure, undergo frequent expert review of procedures, and continue refinement of measures and strategies as required (Cox et al. 2007; Hall 1998). It is fundamental that fishing effort changes are factored into follow-up assessments of mitigation efficacy. For all fishing gear, obtaining estimates of post-release mortality from direct fisheries interactions is an area of research that requires urgent research attention, although monitoring released individuals this is likely to require a large investment.

Despite these challenges, it is crucial that resources are prioritised towards continued development, scientific testing and subsequent implementation and monitoring of proven, effective technical mitigation measures to ensure the ecological sustainability of commercial fisheries. As marine mammal mortality from fishing gear interactions is likely to increase due to human population growth, increasing industrialisation of fisheries, increasing population sizes of some marine mammal species, and fisheries expanding into new areas (Read et al. 2006), improving and implementing effective mitigation is essential. From a global perspective, improving the environmental sustainability of commercial fisheries requires wider dissemination of successful technologies and knowledge of mitigation techniques and comprehensive engagement of fishers in the development of appropriate bycatch solutions (Hall and Mainprize 2005). Developed countries have a level of obligation to assist developing countries in addressing bycatch issues particularly as many marine mammal species have global distributions. At the least, this should entail the publication of research on mitigation design, development, scientific testing of efficacy (or lack of efficacy) and monitoring of operational deployment. It is hoped that this review contributes to this process by having a ‘one-stop-shop’ on the current status of mitigation techniques developed and assessed for marine mammal bycatch in commercial trawl, purse seine, longline, gillnet and pot/trap fisheries.

2.6 Appendices

Appendix 2A: Technical mitigation measures developed to reduce marine mammal bycatch in trawl fishing operations.

General description of trawl fishing and bycatch: Trawling operations involve a net being towed through the water to capture target fish species. Marine mammals are frequently caught in pelagic or midwater trawls as these often target the same pelagic species eaten by marine mammals, have relatively high tow speeds with larger nets, and usually operate within marine mammal diving ranges for extended periods (Fertl and Leatherwood 1997; Hall et al. 2000).

In pair trawl operations, the two vessels may have greater capacity to tow gear faster than single stern trawlers, target fast-swimming fish, and may have high levels of marine mammal bycatch (Northridge et al. 2011; van Marlen 2007). Fishing technology improvements, such as the introduction of factory/freezer vessels, have also expanded operational capacity to areas further offshore potentially increasing the likelihood of interactions with some species (Crespo et al. 1997; Zollett and Rosenberg 2005). The capacity to have larger circumference trawl nets with bigger openings, and rigging parts with greater extension, may increase the bycatch risk of larger animals (Zeeberg et al. 2006), although, contrary to common misconception, factory/freezer vessels do not always use larger nets than smaller trawlers. Conversely, modern pelagic trawlers are usually equipped with state-of-the-art technology, which has increased their ability to selectively target fish (http://www.seafish.org/geardb/gear/pelagic-trawl/).

Marine mammal mortality occurs when individuals enter the net and become trapped, typically when the boat stops 'hauling' and the trawl entrance collapses, or when the net is being shot away and the net is relatively shapeless and slow-moving (Du Fresne et al. 2007; Hamer and Goldsworthy 2006; Tilzey et al. 2006). The majority of marine mammals are caught passively in trawls whereas some (e.g. small cetaceans and fur seals) appear to deliberately enter nets to depredate on captured fish (Fertl and Leatherwood 1997; Hamer and Goldsworthy 2006; Lyle et al. 2016; Wakefield et al. 2017). For some fur seal species, mortality occurs during both net shooting and hauling, as seals are attracted to fish caught in the mesh of the net or to the catch at the end of trawling (Hamer and Goldsworthy 2006). Dolphins may have their rostrums caught in the net while pulling out fish and they sometimes drown when they are caught around the tail stock and have also been caught in turtle exclusion devices (Fertl and Leatherwood 1997). In the UK bass pair trawl fishery, dolphins mostly enter the net to feed on small fish enclosed within the net (Northridge et al. 2005b).

Mitigation measure	Scientific evidence for effectiveness in trawl fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
Net colour		More bottlenose dolphins (*Tursiops truncatus*) were caught in the grey trawl net of one vessel compared to the standard green net of the other vessel (Stephenson and Wells 2006) although this was not formally tested. CAVEATS: ● the two vessels had different management of hydraulic net hauling and so different net speeds through the water during winching could have been a factor in the different bycatch rates of the nets (Stephenson and Wells 2006); ● it is unknown whether different coloured nets may deter some species and, if they do, it is unclear what the mechanism of this would be. As cetaceans and pinnipeds enter nets to feed on captured prey, it seems unlikely that any influence of net colour would be enough to reduce this incentive.	Further investigation required to determine whether net colour could influence bycatch rates.	Not tested properly and so not currently recommended as a mitigation measure.
Net binding	Use of net binding is thought to have reduced fur seal (*Arctocephalus* spp.) bycatch in Tasmania's winter blue grenadier (*Macruronus*	For seabird bycatch mitigation, prior to shooting, sisal string (or similar organic material) is used to bind net sections and prevent net lofting (Sullivan et al. 2004). The tension created by vessel speed stops the mesh opening due to wave and swell action. Once shot-away, the net remains bound on the surface until it sinks and, once trawl doors are paid away and net has sunk beyond diving depth of seabirds, water force	Testing of net binding to assess the efficacy of this technique in reducing entanglements of marine mammals and to establish optimal technical specifications for reducing marine	Although standards have been established for reducing seabird bycatch, there are no specifications

Mitigation measure	Scientific evidence for effectiveness in trawl fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
	novaezelandiae) fishery (Mike Gerner, Australian Fisheries Management Authority (AFMA), pers. comm. 13/09/2016).	moves the doors apart breaking the bindings so the net can obtain its standard operational position. ● No research was identified on the use of binding to reduce marine mammal bycatch. ● Net binding technique would only be effective if trawl net being pulled out of water after each haul (NB: some pelagic trawl operations use fish pumps to transfer the catch directly from codend to vessel hold). ● No reporting of operational details or testing of this technique to reduce marine mammal bycatch. However, net binding has been used in Tasmania's winter blue grenadier fishery, including the additional use of a hydrostatic release mechanism (Mike Gerner, AFMA, pers. comm. 13/09/2016). Net binding was also proposed to reduce fur seal bycatch risk in a recent proposal to introduce pair trawlers to Australia's Small Pelagic Fishery (Atlantis Fisheries Consulting Group 2017).	mammal bycatch. In particular, testing needs to ensure net opens at depths beyond bycatch species diving range.	established for mitigation of marine mammal bycatch.
Exclusion device - additional section of netting inserted between entrance and codend, with angled grid (soft, semi-flexible or hard) that prevents large animals from entering codend and directs them to an escape hole in top or bottom of net; - escape hole provides marine mammals in the back end of a trawl net with an option of escape.	**Hard (usually stainless steel) grid angled upwards to top-opening escape hole:** ● in Australia's blue grenadier fishery, Seal Exclusion Device (SED) with forward-facing, top-opening escape hole had significantly lower fur seal (*Arctocephalus* spp.) bycatch than other SED designs and nets without SEDs - various trials from 2001-2003 (Tilzey et al. 2006). However, Hamer and Goldsworthy (2006) reported no difference in bycatch with SEDs compared to no SEDs, although assessment difficult as very low numbers of seals entered nets during these trials in winter 2003. ● observed New Zealand sea lion (*Phocarctos hookeri*) bycatch rates substantially reduced following widespread use of Sea Lion Exclusion Devices (SLEDs; Figure 2A.1) with top opening, forward-facing escape holes in Auckland Islands midwater arrow squid (*Nototodarus sloanii*,) trawl fishery (Hamilton and Baker 2015a; Thompson et al. 2013b). Review of extensive	CAVEATS: ● rigging and handling large, rigid exclusion devices can be a problem in large pelagic trawl operations; ● difficult to verify performance and efficacy of devices if low interaction rates between marine mammals and nets, and there is often a complex range of factors influencing interactions with net (Hamer and Goldsworthy 2006; Tilzey et al. 2006); ● reduction in observed bycatch rates since SEDs implemented may be, in part, due to reduction in seals entering net rather than SED use (Hamer and Goldsworthy 2006); ● although there is a lack of evidence to show that 'cryptic' mortality is an issue for nets fitted with top-opening exclusion devices (Hamilton and Baker 2015a), there has been some concern that dead animals may fall out of nets or they may obtain injuries that compromise their post-escape survival; ● can be difficult to obtain clear footage from underwater cameras to assess SED/SLED efficacy; e.g. fishing depth (low light, suspended fine debris) and inking from target squid make it infeasible to use video for assessments in Auckland Island squid trawl fishery (Hamilton and Baker 2015a). a) Backward-sloping grid with top-opening escape hole and backward-facing cover: ● significant target fish loss out top-opening hole with backward facing cover (Tilzey et al. 2006); ● in Australia's Pilbara demersal Fish Trawl Fishery (PFTF), there were insufficient numbers of dolphins that interacted with nets to enable assessment of device effectiveness. However, 2/7 bottlenose dolphins exited a top-opening hole. 5/7 observed dolphins backed down into the net, some got stuck by their tail flukes in grid bars and most became exhausted after attempting exit towards net mouth rather than exit out escape hole. One of the 5 dead dolphins was expelled underwater out top-opening hole when the net rotated 180° during the haul so that the hole (with no cover) was orientated downward (Wakefield et al. 2014; Wakefield et al. 2017). ● In U.K. seabass (*Dicentrarchus labrax*) pair trawl fishery, an exclusion grid may reduce common dolphin (*Delphinus delphis*) mortality although it may act as a barrier (i.e. dolphins turn and exit out net mouth) rather than an escape route. Dolphins appeared to detect grid from a distance and attempt to escape in that location rather than finding escape hole at the grid (Northridge et al. 2005a).	● Hard grid exclusion devices with top-opening escape holes need to be adjusted and fine-tuned for each fishery taking into account gear characteristics, operational requirements (e.g. net storage on vessel), towing speed, hydrodynamics of trawl size/grid, escape hole ratios, and physical and behavioural attributes of target and non-target species. ● Urgent research needed to further develop design and test effectiveness of exclusion devices with top-opening escape holes in reducing dolphin bycatch (van Marlen 2007) including: - best location for devices (probably further forward in net), - ensuring escape holes are obvious and easy to escape through, while retaining fish, - better understanding of dolphin behaviour in nets and factors that contribute to dolphin mortality. ● In Australia's blue grenadier fishery, to stop large-sized target fish becoming stuck on a SED grid, an "Acoustic-SED" has specialised split grid with hinged top and fixed lower half. During setting and at fishing depths beyond seals' diving range, the grid's top half is open allowing large-sized target fish to flow into codend.	Exclusion devices with upwardly angled, hard (stainless steel) grids and top-opening escape holes are likely to be effective in reducing pinniped bycatch as well as minimising target catch loss. Forward-facing hood over escape hole further reduces catch losses and the likelihood of dead animals falling out. A kite (e.g. strip of material) and floats on the hood ensures it operates properly and remains open in all conditions (Hamilton and Baker 2015a). Need to specifically refine device components for different fisheries (e.g. grid construction, grid angle, gaps between grid bars, size of escape opening,

Mitigation measure	Scientific evidence for effectiveness in trawl fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
	research to test SLED efficacy concluded that available evidence showed SLEDs greatly increased the likelihood of sea lion survival and that a hood with kite and floats over the escape hole reduced risk of target catch loss as well as ensuring escape hole remained viable for sea lions to exit (Hamilton and Baker 2015a). ● exclusion devices, usually with a metal grid inclined upwards at 45° and a top-opening hatch, are mandatory in relevant krill (*Euphausia superba*) fisheries managed by CCAMLR (Minister for Agriculture; Fisheries and Forestry 2009) and observed bycatch rates of Antarctic fur seals (*Arctocephalus gazella*) has been eliminated (CCAMLR 2017).	b) Backward-sloping grid with top-opening escape hole and forward-facing cover: ● forward-facing cover over escape hole appeared to reduce fish loss although, in some fisheries, there may be issues with grid becoming blocked if target fish are large (Tilzey et al. 2006); ● addition of hood with kite and floats reduced risk of target catch loss and ensured escape hole remained open and viable for sea lion escape (Hamilton and Baker 2015a).	Before hauling, activation of acoustic transponder closes grid so any seals are excluded from codend and can escape via SED top-opening hole (Mike Gerner, AFMA, pers. comm. 13/9/2016). Research assessing the efficacy of the "Acoustic-SED" should be progressed.	construction and elements of hood cover), operationally test and continue to monitor device effectiveness: e.g. grid bar spacing needs to ensure smaller individuals of non-target bycatch cannot pass into codend (Hamilton and Baker 2015a), and escape holes need to be large enough to allow easy escape (e.g. 1.9 m wide for Australian (*Arctocephalus pusillus doriferus*) and long-nosed (*A. forsteri*) fur seals; (Lyle et al. 2016).
	Soft or semi-flexible grids with top-opening escape hole:	Operational configuration on vessels can affect suitability of different SED designs. In Australia's Small Pelagic Fishery, difficult to retrieve and store SED with a hard grid angled upwards to top-opening hole on vessel's on-board net drum storage due to the grid angle being counter to the drum curve. However, a vertical, soft rope mesh grid with top-opening escape hole initially used in this fishery was ineffective as the mesh deformed under the seal's weight, which increased the entanglement risk. Also, the vertical orientation of grid did not direct animals towards the escape hole and there were large losses of target catch (Lyle et al. 2016). In pelagic seabass pair trawl fisheries, it was operationally feasible to handle and manage an exclusion device with a semi-flexible (stiffened polyurethane) grid. However: ● although video footage showed some dolphins were able to identify a top-opening escape and manoeuvre through it, many dolphins (often in weak or catatonic states) were unable to recognise or negotiate out of escape hole. As devices were located far back in net, it was considered most dolphins reaching the grid were unlikely to escape and survive (van Marlen 2007); ● large target fish species (seabass) potentially block the grid; ● grid became easily distorted and fishing crews thought it reduced target fish catches (Bord Iascaigh Mhara and University of St Andrews 2010).	● In Australia's Small Pelagic Fishery, further refinement in SED design is required including - further investigation of top escape opening (with/without hood) to reduce loss of dead seals and target fish, - assessment of unaccounted mortality from animals dropping out, - ensuring seals are directed to escape holes whether or not they actively search for an exit option (Lyle et al. 2016).	

Mitigation measure	Scientific evidence for effectiveness in trawl fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
	Hard grid with bottom-opening: Bottom-opening SED with large escape hole (1.9 m wide for Australian and long-nosed fur seals) reduced seal mortality rates in Australia's Small Pelagic Fishery (Lyle et al. 2016).	In Australia's midwater Small Pelagic Fishery: ● bottom-opening SED with small escape hole had high seal mortality rates compared to bottom-opening SED with large escape hole; ● some dead seals dropped out of bottom-opening escape hole and, without video monitoring, these would not have been included in bycatch reporting (Lyle et al. 2016).		
	Soft or semi-flexible grid with bottom-opening:	In Australia's Pilbara Fish Trawl Fishery (PFTF), to allow retrieval onto net drum, bycatch Reduction Device (BRD) grid was constructed of stainless steel tube and central sections of braided stainless wire with articulated joints at top and bottom. Grid angled down to bottom-opening escape hole with loose netting skirt cover: ● BRD designed to release range of megafauna species; ● bottlenose dolphin bycatch rate reduced with introduction of BRDs in 2006. However, since then, annual fishing effort has declined and dolphin bycatch rates have increased despite mandatory deployment of BRDs (Allen et al. 2014); ● underwater video footage recorded two dolphins that contacted BRD grid, repeatedly pushed against upper surface of net before becoming motionless and falling out bottom-opening escape hole (Jaiteh et al. 2014). Therefore, BRDs may not be as effective in reducing dolphin bycatch as first thought (Allen et al. 2014); ● devices with bottom-opening escape holes thought to be less suitable for air-breathing marine mammals which are more likely to swim upward when trying to escape (Allen et al. 2014); ● only some dolphins with 'trawler-associated' behaviour entered nets and they all swam in same direction as the trawl. If they contacted the grid, they were likely to swim forward and upward and were unlikely to locate and escape from bottom-opening escape hole (Jaiteh et al. 2014); ● devices moved forward from just before codend to start of net extension - prevents dolphins backing into the extension and provides shorter escape route between device and net entrance (Allen et al. 2014); ● target fish lost out bottom-opening escape hole (Stephenson and Wells 2006). In the PFTF, of 705 megafauna individuals, only one bottlenose dolphin (in a top-opening device rotated downwards) and one turtle (in a bottom-opening device) exited a device either dead or in poor condition and, therefore, unaccounted mortality was considered negligible (Wakefield et al. 2014; Wakefield et al. 2017). More recently in Australia’s PFTF, underwater observations from camera footage during 50 day-trawls in Jan/Feb 2013 showed that only four out of 5,908 dolphin interactions (0.07 %) involved a dolphin moving far into the net and presumably interacting with the Bycatch Reduction Device. Of these, two were presumed to exit via the BRD and two occurred within 6 minutes of the net being hauled to the surface and both dolphins being released alive (Santana-Garcon et al. 2018) In European pelagic trawl fleet off Northwest Africa, no cetaceans successfully survived in nets fitted with non-rigid exclusion device with bottom-opening escape		

Mitigation measure	Scientific evidence for effectiveness in trawl fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
		route (designed to release a range of megafauna species). Considered cetaceans unlikely to swim downwards and into a tunnel-like escape route. Need for more research on whether device with a top-opening escape option would be effective in reducing cetacean mortality, although concerns about target fish potentially able to swim out of top escape hole (Zeeberg et al. 2006).		
	Light trawl to catch shrimp with Turtle (and vaquita) Excluder Device - field trials showed reduction in vaquita (*Phocoena sinus*) bycatch - the industrial version of "RS-INP" trawl prototype reduced bycatch-to-shrimp rations between 20-50% without reducing shrimp catch (Aguilar-Ramírez and Rodríquez-Valencia 2010; Senko et al. 2014); - mandatory implementation to replace shrimp gillnets and reduce bycatch risk for vaquita (CIRVA 2012; CIRVA 2014).	A small trawl for shrimp was developed by Mexico's National Institute of Fisheries (INAPESCA) to reduce 'critically endangered' vaquita bycatch in shrimp gillnets. These trawls have several components to improve their environmental performance: (1) turtle excluder device which may allow vaquitas to escape in the rare instances that they are entrapped (CIRVA 2012), (2) bycatch reduction device designed to allow finfish to escape, (3) double rope to avoid damaging seabed, (4) mesh size is progressively reduced along the net, (5) hydrodynamic trawl doors reduce resistance and increase efficiency and, (6) super-light, durable Spectra materials (CIRVA 2014). Also considered that vaquitas avoid boats with engines underway (CIRVA 2012). In 2013, Mexican government mandated use of the small trawl for Upper Gulf of California shrimp fishery. CAVEATS: ● small trawl cannot operate in areas with gillnets; ● fishers reluctant to change gear type; ● small trawl has higher fuel consumption and engine depreciation compared to gillnet operations; ● bycatch of juvenile finfish may be an issue (CIRVA 2014).		Lack of evidence to recommend this measure.
Rope or mesh barriers near net entrance	Lack of scientific evidence for reducing either cetacean or pinniped bycatch.	Rope or mesh barriers near the net mouth are sometimes referred to as an excluder device as they aim to prevent marine mammals entering past the fore section of a trawl net. European/U.K. pelagic seabass pair trawl fisheries: Two net barrier designs showed some potential in allowing dolphins to escape from trawl nets although lack of cetacean interactions during testing meant results largely inconclusive: ● net mesh barrier with top-opening escape holes covered with parallel 'bungee cords' - had video evidence of dolphins escaping but more design refinement and testing required; ● net mesh barrier fitted further forward where trawl net mesh diameter large enough to allow dolphin escape - may reduce dolphin bycatch although results inconclusive and large size of net barrier may increase drag; ● rope and mesh barriers located further forward in net between large mesh and small mesh sections caused unacceptably high levels of gear drag and large reduction in fish catch as target fish were also deterred from entering nets (Bord Iascaigh Mhara and University of St Andrews 2010; Northridge et al. 2005a; van Marlen 2007). Several types of cetacean “barriers” to limit dolphin entry at the net mouth (i.e. vertical ropes in the front part of the trawl) have been under development in the Dutch/Irish pelagic fleet (Zeeberg et al. 2006), although no results of trials have been published.	None.	Lack of evidence to recommend this measure.

Mitigation measure	**Scientific evidence for effectiveness in trawl fisheries**	**Caveats/Notes**	**Research needs**	**Minimum standards/ recommendations**
Acoustic Deterrent Devices (Pingers)	Louder pingers (e.g. Dolphin Dissuasive Device®, particularly model DDD-03H) have been effective in reducing common dolphin bycatch in pair trawl nets (Northridge et al. 2011).	Western Australia Pilbara demersal trawl fishery (PFTF): ● no indication that commercially available *"SaveWave®"* (www.savewave.eu) pingers deter bottlenose dolphins from entering trawl nets; ● trawl operational noise is likely to outweigh any deterrent affect from pingers (Allen et al. 2014); ● 5,908 interactions were recorded using cameras over 50 day-trawls in Jan/Feb 2013, although more than 78% of interactions occurred outside the net. There was insufficient statistical evidence to show that DDDs reduced bottlenose dolphins interactions with the trawl (Santana-Garcon et al. 2018). U.K. seabass pair trawl fishery: ● "*Netmark*" pingers around mouth of trawl did not reduce dolphin bycatch; ● "*Aquamark*" pingers placed in rear of trawl net did not reduce dolphin bycatch - pingers may not be loud enough to deter dolphins and/or dolphins may be too highly motivated to enter net to depredate captured fish (Northridge et al. 2003).	Increase understanding why pingers have resulted in common dolphin bycatch reductions in some studies, e.g. is it a deterrent effect or do pingers enable dolphins to locate an escape route out of the trawl mouth (Berrow et al. 2009). The most effective configuration and spacing of pingers (e.g. "DDDs") for midwater trawls is yet to be determined. Testing of optimal configuration and spacing of loud pingers for midwater trawl operations is required (Northridge et al. 2011). Testing on whether pingers reduce the bycatch of dolphin species (other than common dolphin) in trawl nets.	Operating procedures need to include a program of ensuring DDD-03H pingers are fully operational including batteries being fully charged. According to manufacturer's instructions, the correct number of pingers need to be deployed and located in the correct position on trawl gear (e.g. DDDs should be deployed in at least 10 m of water to effectively broadcast the acoustic signal) (Northridge et al. 2011) .
		Louder pinger devices developed and trialled in European/U.K. seabass pair trawl fisheries (as part of NECESSITY project): "*Cetasaver*" pinger (IFREMER – IX Trawl, France) emits conical direction beam. Deployed on rear part of trawl with beam directed to opening giving 139 dB sound at trawl entrance: ● although trials using *Cetasaver*-03 showed 70% reduction in common dolphin bycatch, sample sizes (number of observations) were not large enough to obtain significant result and more testing required; ● no impact on target bass catch rate was detected; ● *Cetasaver* caused less interference with net monitoring cable (netsonder) compared to DDD pingers which are fixed on trawl wings (see below); ● no acoustic signal has been developed that strongly deters common dolphins in all geographic areas at all times of year; ● background noise of trawl operations is an issue when designing appropriate pingers (Morizur et al. 2007; Morizur et al. 2008). Dolphin Dissuasive Device (DDD) omnidirectional device fixed on trawl wings: ● absence of a sufficient number of control tows (i.e. without pingers deployed) over the period of DDD deployment prevented confidence in results (Northridge et al. 2011). Also, reductions in observed dolphin bycatch may be partly due to decreased fishing effort due to high fuel prices and low target fish availability (de Boer et al. 2012); ● DDDs may degrade and lose charge after 3 years;	Research on impact of widespread use of loud pingers on cetacean behaviour, distribution and ecology. Research impact of pingers on other species in marine environment. Research on potential habituation to pingers for different marine mammal species (Northridge et al. 2011).	

Mitigation measure	Scientific evidence for effectiveness in trawl fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
		● may not be effective if placed in sub-optimal position on trawl gear; ● some concern about impact of widespread use of loud pingers e.g. cetacean displacement from key habitats; ● some concern about cetacean habituation to pingers (Northridge et al. 2011). However, DEFRA (2003) did not consider common dolphin habituation to "DDD" pingers was an issue in the U.K. seabass trawl fishery, although if any decrease in deterrent effectiveness was identified via on-board observer monitoring of deployed pingers, it may be difficult to link to habituation effects. In controlled exposure experiments (compared to noisy, complex conditions around trawl operations), there was a lack of consistency in behavioural response of common dolphins to *Cetasaver*-03 and DDD-02F pingers (Berrow et al. 2009). The positioning of pingers should ensure there is no impact on operational electronic equipment. Due to beam geometry, quieter, directional "Cetasaver" devices set on rear parts of the trawl (although exact location was not defined) caused less interference with the net monitoring system compared to omnidirectional "DDD" devices fixed on trawl wings and, therefore, skippers preferred to trial "Cetasaver" devices (Morizur et al. 2007). AquaTech 363 Interactive Pinger: - designed to emit wide band deterrent signals only in response to echolocations from common dolphins, - could prevent habituation to continuous signal, reduce noise pollution and require less power supply, - initial tests on captive bottlenose dolphins in Sweden and in direct playback experiments with bow riding bottlenose dolphins in Ireland showed device consistently responded to echolocations and dolphins exhibited evasive behaviour. However, subsequent trials showed device not effective in deterring common dolphins (van Marlen 2007); - trialled different deterrent signals (e.g. killer whale (*Orcinus orca*) vocalisations) but a consistently effective signal has not been identified (Bord Iascaigh Mhara and University of St Andrews 2010). The effectiveness of pingers in reducing dolphin bycatch in the Dutch/Irish pelagic trawl fleet off Mauritania was investigated (Zeeberg et al. 2006), but there has been no further reporting on this work. DDDs are deployed in New Zealand jack mackerel trawl fisheries and are considered to be effective in deterring common dolphins although there is no formal testing of this (Deepwater Group 2018).		
Auto-trawl systems	The use of otter-board sensors and eliminating sharp turns while trawling appear to have reduced bottlenose dolphin mortalities (from observer data) in the Australia Pilbara demersal	Auto-trawl systems, developed to improve fishing efficiency, help maintain the shape of trawl nets (particularly when turning) via telemetry and sensors on otter boards and the net. Auto-trawl systems use auto-tensioning winch systems (e.g. RAPP® Hydema, Hydraulik Brattvaag (Rolls-Royce®) systems) and, to a lesser extent, net monitoring systems to maintain the trawl shape through monitoring and controlling the trawl doors via telemetry and sensors (e.g. Marport True Trawl geometry system). Ensuring the net entrance does not collapse during trawling operations may reduce marine mammal	More scientific testing is required to evaluate auto-trawl systems as a marine mammal bycatch mitigation measure.	Yet to be established as a mitigation measure.

Mitigation measure	Scientific evidence for effectiveness in trawl fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
	fish trawl fishery from 12.6 to 5.2 dolphins per 1000 trawls (Allen et al. 2014; Wakefield et al. 2014; Wakefield et al. 2017).	entrapment in trawl nets and may help maintain effective operation of excluder devices. However, note that bycatch risk is not always greatest during setting, hauling or turning. In Australia's Small Pelagic Fishery, most seal interactions occurred during the fishing phase (depths usually <150 m) and seals were not considered to be particularly vulnerable during turning and hauling when net geometry was potentially compromised (Lyle et al. 2016).		

Appendix 2B: Technical mitigation measures developed to reduce marine mammal bycatch in purse seine fishing operations.

General description of purse seine fishing and bycatch: In purse seine operations a large circular 'wall' of net set around a fish shoal is 'pursed' together at the bottom before hauling to stop fish escaping downwards under the net (Hall and Roman 2013). Sets are categorised on the different methods that target fish are detected and encircled. These include 'School' sets where target fish are detected from activity at or near the water surface, 'Dolphin' or 'Porpoise' sets where small cetaceans associated with tuna schools are encircled along with targeted fish, and 'Floating Object' sets where sets are made around pelagic fish that naturally congregate around natural or artificial floating objects. 'Fish Aggregating Devices (FAD)' sets are sets around artificial floating elements (e.g. ropes, netting material) that are attractive to pelagic fish, and which include a relocation aid (e.g. radar reflector, radio buoy, satellite tracker) (Hall and Roman 2013). Although marine mammals encircled in nets may escape during net pursing by diving under the net or through the opening before net ends are fully together, some species may not identify escape options and there are no clear escape routes once pursing is complete (Hall and Roman 2013). Marine mammals are particularly susceptible to mortality in pelagic operations when fisheries deliberately set on tuna schools associated with dolphins, although accidental entrapment also occurs (Gilman 2011; Hall and Roman 2013). Although interactions with whales are rare (Amandè et al. 2011), some fisheries have recorded whale injuries or deaths (Anderson 2014; Molony 2005; Romanov 2002). Seal and sea lion bycatch may occur (David and Wickens 2003; Hückstädt and Krautz 2004), while pinnipeds may also impact operations by eating target catch (David and Wickens 2003; Hückstädt and Antezana 2003) or cause encircled fish to escape before net closure (Shaughnessy et al. 2003).

A shift from 'Dolphin sets' to sets around FADs (Hall and Roman 2013; Maufroy et al. 2017) has raised new environmental concerns regarding overfishing and marine species' entanglement in FAD components (Fonteneau et al. 2000; Hall and Roman 2013; Restrepo et al. 2014). Most technical mitigation development and research currently focuses on improving FAD design to reduce shark and turtle entanglement (Delgado de Molina et al. 2005; Goujon et al. 2012; Restrepo and Dagorn 2011; Restrepo et al. 2014; Restrepo et al. 2016).

Mitigation measure	Scientific evidence for effectiveness in purse seine fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
"Dolphin gate" and cork-line weights		Dolphin gate: detachable section of cork-line which, when removed, allows section of net to sink creating opening for encircled dolphins to exit. Cork-line weights: purpose-built attachable weights to help sink cork-line. As there was a lack of evidence to show that these gear modifications reduced short-beaked common dolphin bycatch (Hamer et al. 2008), they are no longer included in Code of Practice for South Australian Sardine Fishery (Ward et al. 2015).	More refinement and testing is required to show whether these gear modifications are effective (Hamer et al. 2008).	Not yet confirmed as an effective mitigation option.
"Back-down" manoeuvre with addition of **"Medina" panels** and use of small boats to encourage escape of marine mammals	Following widespread use of 'back-down' manoeuvre, there was a 97% reduction in annual dolphin bycatch between 1986 and 1995 in Eastern Pacific Ocean (EPO) tuna (*Thunnus* spp.) fisheries (Hall and Roman 2013; Hall 1996; Restrepo et al. 2016). The addition of smaller mesh "Medina" panels in 'back-down' area of net reduces risk of dolphin entanglement (National Research Council 1992).	"Back-down" manoeuvre: a reverse manoeuvre of the fishing vessel which causes a section of cork-line furthest from the vessel to submerge thereby enabling encircled dolphins to actively escape over the net (Hall and Roman 2013). "Medina" panel: panels of relatively small mesh netting (50 mm or less) sewn into the net and surrounding back-down area where small cetaceans most likely to contact the net. Panels are usually one or two strips deep and 330 m long (FAO 2017). PROS: ● Back-down manoeuvre does not appear to negatively impact target catches (Hall 1996) with trials showing little risk of tuna escapement (Restrepo et al. 2016). CAVEATS: ● Unclear whether the reduction in dolphin mortalities in EPO fisheries is directly linked to use of back-down manoeuvre and Medina panels as it is likely to be due to a range of operational changes (Hall and Roman 2013; Hall 1996; Restrepo et al. 2016). ● Despite large reduction in observed bycatch levels, populations of northeastern offshore spotted dolphin (*Stenella attenuata attenuata*) and eastern spinner dolphin (*S. longirostris orientalis*) have not shown clear signs of recovery possibly due to	None suggested. U.S. Marine Mammal Protection Act 1972 - helped to ensure dolphins released from nets via increased regulation, observers on fishing vessels and gear inspections (Gerrodette and Forcada 2005). Research is required on post-encirclement and post-release survival of bycatch animals (Restrepo et al. 2014).	The addition of Medina panels should be used in conjunction with back-down manoeuvre to reduce risk of dolphin entanglement if animals make contact with net on exit.

Mitigation measure	Scientific evidence for effectiveness in purse seine fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
		environmental and ecosystem changes; the negative effects of chasing and encirclement on survival and/or reproduction; and the under-reporting of ‘cryptic’ bycatch mortality (due to lack of observers on smaller vessels, some dead animals being unrecorded and post-escape mortality of injured animals) (Gerrodette and Forcada 2005). - Frequent chasing and encirclement of cetaceans may impact reproductive success including reduced reproductive rates from increased stress, and loss of calves during the chase (Archer et al. 2004; Archer et al. 2001; Cramer et al. 2008; Edwards 2006; Gerrodette and Forcada 2005). - Population modelling indicated populations could be affected by potential cryptic effects from interactions with purse seine operations (Wade et al. 2007). - These dolphin populations have shown slight increases in numbers in more recent (2000s) surveys (Cramer et al. 2008). ● Common dolphins released from Chilean industrial purse seine nets sustained severe face wounds, had difficulty swimming and post-escape survival rates were unknown (González-But and Sepúlveda 2016). ● Although captured whales are usually large enough to break out of nets and it is often considered they suffer no direct mortality from purse seine operations (Amandè et al. 2011), their post-escape survival may be compromised from entanglements in netting pieces or other injuries incurred (Anderson 2014). No studies were identified that have monitored the post-release survival of marine mammals, although there has been post-release monitoring of sharks (Eddy et al. 2016; Escalle et al. 2016; Hutchinson et al. 2015).		

Appendix 2C: Technical mitigation measures developed to reduce marine mammal bycatch in longline fishing operations.

General description of longline fishing and bycatch: Longlining operations aim to catch target fish using a line with a series of baited hooks set in the water column. Longline gear can be set on the seabed (demersal longlining), floated off the bottom at various fishing depths (semi-pelagic longlining) or suspended from floats at the surface (pelagic longlining). Compared to demersal fisheries, pelagic fisheries typically use longer snoods (i.e. the short branch line attached to the main line with the hook attached to the other end), have multiple buoys at the surface and use whole baits.

While marine mammal interactions with longline fishing operations predominantly results in depredation of the bait or hooked catch, depredation behaviour also puts marine mammals at risk of becoming hooked or entangled resulting in injury or mortality (Bigelow et al. 2012). Demersal longline fisheries have reported depredation by odontocetes mainly during the haul when hooked fish are easier to access than during the soak (Ashford et al. 1996; Gilman et al. 2006; Goetz et al. 2011; Guinet et al. 2015; Hamer et al. 2012; Nolan et al. 2000; O'Connell et al. 2015; Soffker et al. 2015; Tixier et al. 2014). However, interactions between southern elephant seals (*Mirounga leonina*) and a sub-Antarctic demersal longline fishery have been recorded during the line soak period at depths >1 km (van den Hoff et al. 2017). In pelagic longline fisheries, depredation by odontocetes is known to occur during all stages of the fishing process (Hamer et al. 2015; Passadore et al. 2015a; Passadore et al. 2015b; Rabearisoa et al. 2015; Rabearisoa et al. 2012). Marine mammal bycatch has been quantified for some longline fisheries (Passadore et al. 2015a), although the rates of both depredation and bycatch in longline fisheries is likely to be underestimated (Hamer et al. 2012). This is partly due to the difficulties of observing interactions at all stages of fishing when longlines are often many kilometres long and set at great depths, particularly in demersal operations (van den Hoff et al. 2017). Longline bycatch has been identified as a threat to some populations; e.g. the Hawaiian Island pelagic population of false killer whales *Pseudorca crassidens* (Baird et al. 2015; Bigelow et al. 2012; Carretta et al. 2016).

Mitigation measure	Scientific evidence for effectiveness in longline fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
Acoustic methods - active	No studies have shown that acoustic devices are consistently effective in deterring cetaceans from an active longline fishing area.	Acoustic Harassment Devices (AHD), usually >180 dB, and Acoustic Deterrent Devices (ADD), usually < 180 dB, aim to force depredating odontocetes away from a fishing area and Echolocation Disruption Devices (EDD) are designed to prevent odontocetes from locating a hooked fish. ● Eating hooked fish is likely to be a strong incentive driving most marine mammal interactions with longlines, and, therefore, the use of acoustic devices is unlikely to substantially reduce interactions; ● Although ADDs and AHDs have been used in some fisheries, there has generally been a lack of replicated controlled experimental trials to determine their efficacy in deterring marine mammals; ● When testing the device efficacy , it is difficult to isolate whether device deployment is the variable responsible for any measured reduction in marine mammal interaction rates (Hamer et al. 2012). "Longline Saver", up to 182 dB (Mooney et al. 2009), and an AHD, 195 dB (Tixier et al. 2015): ● ineffective at deterring depredating killer whales; ● killer whales only had significant response to initial exposure to device activation; ● killer whales appeared to habituate to noise during subsequent exposure and had insignificant behavioural response to activation of AHD after 3rd exposure, ● concern about harmful disturbance to toothed whale echolocation capabilities if they regularly attend longliners equipped with loud devices (Mooney et al. 2009; Tixier et al. 2015).	None.	Currently no evidence to recommend this measure.

Mitigation measure	Scientific evidence for effectiveness in longline fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
Acoustic methods - passive	Indications that more target sablefish (*Anoplopoma fimbria*) were caught on beaded gear as well as a decrease in depredation events by sperm whales (*Physeter microcephalus*) on beaded gear compared with control longlines, although results inconclusive(O'Connell et al. 2015).	Passive acoustic deterrents aim to reduce whale depredation by confusing whale echolocation abilities and/or masking the presence of a hooked fish. Acrylic spherical beads (25 mm diameter) were evaluated to have similar 'acoustic return' to sablefish (the target species) and were attached to each ganglion with the aim of confusing sperm whales attempts at identifying the presence of a sablefish on a hook: ● no difference between fishing gear with device (bead) or without although this result primarily due to lack of power in the at-sea experiment, ● operational testing proved difficult due to normal variability between vessels, difficulty in increasing sample size, and lack of predictability of depredation events (O'Connell et al. 2015).	To date, only tested in pelagic fisheries. More testing required, with sufficient statistical power, to determine efficacy of this technique.	Currently no evidence to recommend this measure.
Weak hooks	There was some indication from at-sea trials that weak hooks have the potential to reduce bycatch of marine mammals in pelagic longlines, although this result is not statistically significant (Bayse and Kerstetter 2010). Laboratory trials showed that weak hooks could help reduce damage to soft lip tissues and reduce potential for complete hooking a marine mammal's lower jaw (McLellan et al. 2015).	“Weak” hooks, constructed of thinner gauge wire, are designed so that they straighten under pressure exerted when a large organism is hooked to allow release of bycatch species. For “weak” hooks to be a bycatch mitigation option in a fishery, there needs to be a size difference between the larger bycatch species and relatively smaller target species (Bayse and Kerstetter 2010). Circle (weaker) hooks are a standard requirement to reduce bycatch of a range of species in many commercial pelagic longline fisheries in the US (Richards et al. 2012), although there is no clear evidence that they are effective in allowing caught cetaceans to straighten hooks and escape. For example, the deep-set Hawaii longline fishery is required to use circle hooks with a maximum wire diameter of 4.5 mm and an offset of 10 degrees or less (Clarke et al. 2014). PROS: ● no significant reduction in total tuna catch or of any individual target species catch, ● weak hooks had higher catch per unit effort (CPUE) for swordfish (*Xiphias gladius*) and tuna (Bayse and Kerstetter 2010); ● a study in the Gulf of Mexico showed catches of spawning bluefin tuna (the bycatch species of concern) were statistically significantly reduced with the use of weak hooks although yellowfin tuna catch rates were the same (Bigelow et al. 2012). ● straightening rates of weak hooks has varied between studies with 0.291 per 1000 hooks in the Hawaii deep-set longline fishery study (Bigelow et al. 2012) and 0.439 for weak hooks in the yellowfin tuna experiment in the western Atlantic (Bayse and Kerstetter 2010). CONS/CAVEATS: ● in a western North Atlantic Ocean pelagic fishery, there was no statistical reduction in catch rates of bycatch species (predominantly marlin and shark species) with use of "weak" hooks, and only one observation of a false killer whale being released and that was from a 'strong' circle hook, ● unknown level of post-release mortality for marine mammals that retain a hook and/or line, ● do not address the issue of depredation or reduce interaction levels (Bayse and Kerstetter 2010); ● assessment of bycatch reduction (sea turtle as well as cetaceans) using circle (weaker) hooks and differing baiting techniques as not possible due to low rates of interactions (Richards et al. 2012).	To date, only tested in pelagic fisheries and more research is required with adequate statistical power. Requires more at-sea trials in different fisheries. Need to assess post-release survival of marine mammals that have been hooked.	Currently not enough evidence to recommend this measure.

Mitigation measure	Scientific evidence for effectiveness in longline fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
Catch protection devices - demersal longline	Chilean net sleeves or 'cachaloteras' (Figure 2C.1): Reduced toothed whale interactions and depredation in a Chilean fishery for Patagonian toothfish (*Dissostichus eleginoides*) from maximum of 5% of total catch in 2002 (no net sleeves) to 0.36% in 2006 with net sleeves (Moreno et al. 2008). 'Umbrella and stones' net sleeves (Figure 2C.2): There were indications that net sleeves reduced sperm whale depredation and cetacean bycatch risk in a Spanish longline fishery, although no statistically significant results were reported due to low interaction rates, i.e. small sample sizes (Goetz et al. 2011).	Chilean longline gear, a modification of the traditional Spanish double line system, has a main line weighted at each end and secondary branchlines containing several hooks also with weights, resulting in faster baited hook sink rates. "Cachaloteras" or net sleeves are typically windsock-shaped sleeves that slide down over hooks and captured fish during hauling. The Chilean system configuration has the hooks directly above the weights ensuring a rapid sink rate. Net sleeves are durable and can be used for long time although are a substantial initial cost (Moreno et al. 2008). CAVEAT: In Chilean fishery, cachaloteras did not adversely affect target fish catch with Catch Per Unit Effort (CPUE) higher in 2006 with net sleeves than in 3 of the prior 4 years (Moreno et al. 2008) whereas 'umbrella and stones' net sleeves significantly reduced toothfish catch rates in Spanish fishery in Southwest Atlantic (Goetz et al. 2011). In the Chilean system, over time, killer whales have learnt to depredate around cachaloteras although the devices are still highly effective for reducing sperm whale depredation rates (Arangio 2012).	Relatively new system that should be monitored and possibly refined further. Need to test broader applicability of net sleeves and further testing of impact on target fish catch rates. Arangio (2012) identified that there is a project (by C.A. Moreno) on a modified cachalotera to prevent killer whale depredation by completely encapsulating the hooked toothfish.	No global standards yet.
Triggered catch protection devices - pelagic longline Devices designed to be triggered via a line tension mechanism when a fish is caught and to descend over the hooked fish.	"Chain device" and "Cage device" (Figure 2C.3): There were indications that these devices may deter depredating toothed whales and reduce bycatch risk for the whales, although the number of interactions during trials was too low (n=4) to obtain significant results (Hamer et al. 2015).	"Chain device" (two 1500 mm stainless steel chains) designed to simulate line tangles and, therefore, draw on any negative previous experience of a whale's temporary entanglement in fishing gear. "Cage device" (cone-like structure made of loops of monofilament nylon, lengths of branchline nylon and joined with aluminium swages) physically covered hooked fish with a barrier. For both devices, branchline routed through a "dog-leg" on the activating mechanism so that device triggered when tension on the hook (from a caught fish) straightened the branchline. PROS: ● impact on target fish catch rates, size, and survival was negligible. CONS: ● post-release fate of whales is unknown, ● extra crew member required to handle devices during setting and hauling, ● hauling time was increased with devices, ● cost of each device and cost of retro-fitting is high and only viable if significant increase in economic benefits from reduced depredation - increased uptake of devices would also enable mass production and reduce overall costs, ● devices did not always deploy when fish hooked (Hamer et al. 2015).	To determine efficacy of this technique, more testing is required to verify preliminary results, and design improvements are required to improve reliability of devices (Hamer et al. 2015).	No global standards yet.
	"Spiders" and "socks": Depredation rate on sets with devices was not significantly	"Spiders" - 8 polyester 'legs' maintained in shape by a 100 mm diameter plastic disc. "Socks" - conical net made of fibreglass mosquito netting or propylene fibre net.	To obtain more robust results, need larger sample sizes and tests need to be carried out in operations with	

Mitigation measure	Scientific evidence for effectiveness in longline fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
	different to those without devices, although relatively low rates of whale interactions observed during the at-sea testing. This initial work showed that these devices have the potential to reduce toothed whale depredation (Rabearisoa et al. 2012).	PROS: ● no increased sink time of the branchline with device attached, ● CPUE not significantly reduced by use of devices, ● devices able to be re-used increasing the cost effectiveness of this mitigation', ● spiders had a lower entanglement rate (3.6%) with fishing gear than socks which had a high entanglement rate of 17.8%. CONS: ● some devices were triggered in absence of a caught fish and some did not trigger when a fish was caught, ● increased time required during setting to attach device onto branchline, ● devices significantly increased hauling time due to resistance in the water which also resulted in entanglements of branchline or around the mainline (Rabearisoa et al. 2012).	higher depredation rates (Rabearisoa et al. 2012).	
	"DEPRED" device (Figure 2C.4): Initial at-sea trial assessed small delphinids interactions and depredation of small pelagic fish (SPF) with or without protection from a device and assumed behaviour would be similar for larger toothed whales interactions with large fish (tunas, billfish). More than twice as many unprotected SPF were damaged by dolphins compared to the SPF protected by DEPRED devices (Rabearisoa et al. 2015).	"DEPRED" device: eight 1 m long streamers made of tarpaulin material and fixed on a 2 cm diameter PVC tube. Bottom streamers weighted to act as physical barrier to protect hooked fish from depredation and upper streamers float around to deter predators from approaching hooked fish. Cons: ● potential that some bottlenose dolphins became habituated to the devices, ● unknown if this device has any effect on target catch rates (Rabearisoa et al. 2015).	Only tested using dolphins (as a proxy for pilot whales (*Globicephala macrorhynchus*) and killer whales) and SPF (as a proxy for tuna/swordfish). Requires further testing and device development to include trigger mechanism (Rabearisoa et al. 2015).	

Appendix 2D: Technical mitigation measures developed to reduce marine mammal bycatch in gillnet fishing operations.

General description of gillnet fishing and bycatch: Gillnets passively catch target species using a wall of virtually invisible netting typically constructed of monofilament or multifilament nylon. Mesh sizes are designed for the head of target fish to pass through and become entangled, usually around the gills. The variations in gillnet design are well described by (He 2006b). Gillnets include:

- Set gillnets (or 'setnets') which are usually used in coastal waters and are anchored at each end with bottom weights and floats at the top (bottom set gillnets are anchored to the sea bed); and,
- Drift gillnets (or 'driftnets') which are generally accompanied by the vessel and held at the required fishing depth using a system of weights and buoys attached to the headrope, footrope or floatline.

Other related fishing gear includes tangle nets which have larger meshes and little or no floatation and sit on or close to the sea bed targeting larger species (e.g. flathead); and trammel nets which catch fish in pockets of light, small mesh sections hung between large mesh nets.

Gillnets incidentally entangle a wide range of marine mammals which may drown when caught in set gillnets or, if entangled in drift gillnets, may drag gear with them which can impact feeding ability, constrict growth and/or cause infection. Marine mammal interactions may involve associated lines as well as the net itself (He 2006b). Bycatch rates may be affected by wind and/or weather, net setting depth, time of year or season, soak duration and gear design including mesh size, net size, hanging ratio, twine diameter and type (Northridge et al. 2017) and correct net setting protocols are crucial (He 2006a; He 2006b). Additional cryptic or unrecorded mortality may include dead animals dropping out of nets as well as post-escape mortalities from injuries or partial entanglement (Hamer et al. 2011; Uhlmann and Broadhurst 2015).

Globally, over the last two decades, at least 75% of toothed whale, 64% of baleen whale and 66% of pinniped species have been recorded as bycatch in gillnet fisheries (Reeves et al. 2013). Gillnet bycatch is identified as one of the most serious threats to small cetacean (Kraus et al. 1997; Prajith et al. 2014; Read et al. 2006) and pinniped (Hamer et al. 2013) populations. Understanding the behaviour of marine mammals and the mechanics of how and why they interact with nets is important for developing solutions to reduce bycatch. It is thought that marine mammals have problems both with detecting and avoiding nets (Lien et al. 1995). Pinnipeds probably rely on vision underwater and, particularly in low light, are unlikely to detect gillnet monofilaments other than at close range (Martin and Crawford 2015). Echolocating cetaceans may become entangled if they are not using their sonar and so may not detect a net's presence, they may not identify echoes from a net as something significant, or may have difficulty in echolocating nylon monofilament netting which has a similar density to seawater (Dawson 1991). Recent research confirmed that gas embolism occurs in turtles following release from both gillnets and trawls, which can lead to the post-release mortality of some individuals (Fahlman et al. 2017) and this may also impact other breath-holding diving vertebrates, including pinnipeds and cetaceans, retrieved from depths (Moore et al. 2009).

There has been substantial research on the development, testing and deployment of pingers to reduce small cetacean bycatch in gillnets. This updated assessment draws on a number of previous mitigation reviews with varying objectives and focuses (Childerhouse et al. 2013; Dawson et al. 2013; Leaper and Calderan 2018; Mackay and Knuckey 2013; Northridge et al. 2017; Read 2008; Read 2013; Reeves et al. 2013; Uhlmann and Broadhurst 2015; Waugh et al. 2011) as well as other studies from peer-review literature and 'grey' literature.

Mitigation measure	Scientific evidence for effectiveness in gillnet fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
Acoustic Deterrent Devices (Pingers)	**Dukane Netmark™ 1000** pingers: ● significantly reduced **harbour porpoise** (*Phoecoena phocoena*) bycatch in a wide range of studies - see Table 2 and Table 3 in Dawson et al. (2013), e.g. (Berggren et al. 2002; Gönener and Bilgin 2009; Kraus et al. 1997; Palka et al. 2008; Trippel et al. 1999). ● significantly reduced **common dolphin** bycatch rates and eliminated bycatch of **beaked whales** (*Berardius bairdii*, *Mesoplodon* spp., *Ziphius* spp.) in U.S. shark setnets (Barlow and Cameron 2003; Carretta and Barlow 2011; Carretta et al. 2008). ● significantly reduced bycatch of **Franciscana dolphin** (*Pontoporia blainvillei*) in Argentina (Bordino et al. 2002). ● significantly reduced small cetacean (including **bottlenose dolphins**, **common dolphins** and p**ilot whales**) bycatch in Peru's small-scale driftnet fishery (Mangel et al. 2013).	Pingers are small battery-operated, electronic devices that actively emit a sound to deter non-target animals from operational fishing areas. They differ in pulse duration, sound level and pulse direction (directional or omnidirectional) with relatively low output source levels (<160 dB) in the 10-100 kHz frequency range (Dawson et al. 2013; Mackay and Knuckey 2013). Dukane Netmark™ 1000 pingers are commercially available and have a fundamental frequency of 10 kHz (range 10-12 kHz) and source level of 132 dB (range 120-146 dB). Ineffective response/non-significant result: ● Inconclusive whether pingers effective in reducing **Hector's dolphin** (*Cephalorhynchus hectori*) entanglements due to: - pingers used in combination with other mitigation measures and so isolating their effect is not possible, - there is insufficient observer coverage to determine if any measure is effective (Dawson and Slooten 2005), - a study that reported a significant response from Hector's dolphins to pingers (Stone et al. 1997) was not statistically valid (Dawson and Lusseau 2005). ● Although indications that pingers reduced **tucuxi** (*Sotalia fluviatilis*) bycatch in Brazil gillnet fisheries (Monteiro-Neto et al. 2004), a lack of statistical rigour (i.e. pseudoreplication) was identified in this study which reduced the ability to conclude the reduction in bycatch was significant (Dawson and Lusseau 2005). Dawson and Lusseau (2005) suggested alternative approaches for improving the statistical rigour of these studies including using only one observation per dolphin group in analyses or undertaking analyses using a comparative modelling approach. ● No **harbour porpoise** were caught in either control or experimental (with pingers) gillnets in the Swedish Skagerrak Sea, possibly due to displacement of porpoise, high prey availability in other areas (herring (*Clupea harengus*) appeared to be more abundant in other areas), or pingers acting as passive reflectors (Carlstrom et al. 2002); ● Pingers did not deter **bottlenose dolphins** from gillnets (Cox et al. 2003). ● Pingers did not reduce **harbour seal** (*Phoca vitulina*) bycatch (Gearin et al. 2000). ● When assessing the efficacy of pinger deployment over a period of time (e.g. 16 year program), it is important to consider whether bycatch reduction is due to pingers or if any decrease in marine mammal abundance confounds the results (Carretta et al. 2008). Varied results on the impact of pingers on herring, an important prey item for harbour porpoises: - in Gulf of Maine, no difference in catches for some fish species whereas herring were captured more in control than active strings indicating that this may have been a factor in the decrease in harbour porpoise bycatch (Kraus et al. 1997). - in Bay of Fundy, pinger use resulted in a significant reduction in harbour porpoise bycatch and no difference in catch rates of herring, Atlantic cod (*Gadus morhua*), and pollock (*Pollachius virens*) (Trippel et al. 1999). Culik et al. (2001) also found no difference in herring catches with use of pingers. PROS: ● Pingers caused no adverse effects on target fish catches in the Gulf of Maine (Kraus	General research needs for pingers: Further research to understand the behaviour of cetaceans in response to 'reactive pingers' (Leeney et al. 2007). It is important that ongoing monitoring is implemented to ensure the effectiveness of pingers deployed in an operational fishery is maintained. As pingers rely on changing animal behaviour to avoid bycatch, they should only be implemented in a fishery after rigorous testing on all likely bycatch species (Hodgson et al. 2007). Test the efficiency of acoustic deterrents with different frequencies (which may not attract sea lions) in reducing Franciscana bycatch in small fishing communities.	In England, mandatory use of approved pingers on >12m vessels since July 2013 (Crosby et al. 2013). In the California-Oregon drift gillnet fishery, pingers have been mandatory since 1998. Pinger specifications are: - frequency of 10 kHz (± 2 kHz) at 132 dB re 1µPa @ 1 m (± 4 dB), - staggered spacing between the floatline and leadline of net and with a maximum spacing of 91.44 m (300 ft) on each line (Mackay and Knuckey 2013).

Mitigation measure	Scientific evidence for effectiveness in gillnet fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
		et al. 1997), in U.S. chinook salmon (*Oncorhynchus tshawytscha*) gillnets (Gearin et al. 2000), in Swedish bottom-set gillnets (Carlstrom et al. 2002), in U.S. shark drift setnets (Barlow and Cameron 2003), in Turkish Black Sea turbot (*Schophthalmus maeoticus*) fisheries (Gönener and Bilgin 2009), or in Argentinian bottom gillnet fisheries (Bordino et al. 2002); ● In the Gulf of Maine, no increase in damage to catch caused by seal depredation with the use of pingers (Kraus et al. 1997). ● No evidence of harbour porpoise (Dawson et al. 2013; Palka et al. 2008), common dolphin or beaked whale (Carretta and Barlow 2011) habituation in fisheries with long-term (e.g. 14 year) pinger use. However, observational studies have shown evidence of habituation in an experimental context; e.g. Cox et al. (2001). CAVEATS/CONS: ● Unclear why porpoises avoided nets with pingers. If deterred because prey species (e.g. non-target herring) avoided pingers, then pingers may be less effective in reducing porpoise bycatch in areas where herring not a porpoise prey item (Dawson et al. 1998). ● Cost of pingers and batteries may be substantial and limit their use in some fisheries, particularly in developing countries (Dawson et al. 2013; Dawson et al. 1998). ● Concern about habituation to pingers - harbour porpoise echolocation and movements around a mooring with pinger showed initial displacement of 208 m, but this diminished by 50% within 4 days (Cox et al. 2001). ● Effect of pingers greatly reduced if not all deployed pingers are functional. ● Even with pingers deployed correctly, harbour porpoise bycatch reduction appears to have been less effective in operational fisheries (50-80% reduced bycatch) than during original scientific testing (92% in Kraus et al. (1997)) (Orphanides and Palka 2013). ● In Gulf of Maine and Bay of Fundy commercial bottom-set gillnet fisheries, harbour porpoise bycatch was reduced to below Potential Biological Removals (PBR) levels from 1999-2003. Since then, bycatch has been above the PBR level in most years, primarily due to poor compliance with pinger requirements, as well as changes in fishing effort and fisheries distribution (Orphanides and Palka 2013). However, these authors consider that the effectiveness of pingers in mitigating bycatch could be improved with increased adherence to management requirements; e.g. ensuring pingers are functioning correctly and with the required number in the correct net location. ● Potential of pingers causing 'dinner bell' affect for some pinniped species - initial testing showed **California sea lion** (*Zalophus californianus*) bycatch reduced with pinger use (Barlow and Cameron 2003). However, monitoring of pinger deployment over 14 years subsequently showed sets with pingers had almost twice the amount of sea lion bycatch although this increase most likely due to increased sea lion abundance and not pinger use (Carretta and Barlow 2011). ● Potential exclusion from habitat from pinger use - Modelling of a harbour porpoise population in inner Danish waters indicated that, while widespread pinger deployment eliminated bycatch, the consequential avoidance of high quality foraging habitat negatively impacted individual survival and total population size (van Beest et al. 2017). Pingers also resulted in increased interactions between fishing gear and **South American sea lions** (*Otaria flavescens*) (Bordino et al. 2002). To reduce common and		

Mitigation measure	Scientific evidence for effectiveness in gillnet fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
		bottlenose dolphin bycatch while ensuring pingers do not increase Australian sea lion (*Neophoca cinerea*) interactions with gillnets, pingers within the auditory range of sea lions should be avoided (Mackay and Knuckey 2013).		
	Lien model: ● significantly reduced **harbour porpoise** bycatch (Culik et al. 2001; Gearin et al. 2000).	Base frequency of 2.7 kHz and harmonic up to 19 kHz with 115 dB source level. ● no difference in herring catches with use of pingers (Culik et al. 2001).		
	Dolphin Dissuasive Device® (DDD-03H®): ● significantly reduced **harbour porpoise** bycatch in offshore set gillnet and tangle/trammel net fisheries (Northridge et al. 2011). Due to low interaction rates, the effect of these pingers on **common dolphin** bycatch in gillnets is inconclusive (Bord Iascaigh Mhara and University of St Andrews 2010).	5-500 kHz with 165 dB re 1 Pa @ 1m PROS: ● As DDD pingers were louder, fewer were needed and they were deployed on end ropes of fleet rather than on nets so less prone to damage. ● Most harbour porpoises caught in nets using DDDs were >2 km from nearest device. CONS: ● It is inconclusive whether DDD pingers are effective in deterring **common dolphins** and **seals**. - Interaction rates with common dolphins were low and so samples sizes too small to evaluate if DDDs effective in reducing dolphin bycatch; - Although appeared that seal, mainly **grey seal** (*Halichoerus grypus*), bycatch was less in nets with DDDs, the results were misleading due to single unusual trip that caught 19 seals (Northridge et al. 2011).		
	BASA pingers (used in Australia)	Either 4 kHz or 10 kHz pingers with about 133 dB sound level. ● Pingers unlikely to be effective in deterring **dugongs** (*Dugong dugon*) or in reducing dugong mortalities in fishing nets (Hodgson et al. 2007).		
	AQUAmark® 100 pingers: ● significantly reduced interactions of **harbour porpoise** with gillnets (Hardy et al. 2012; Larsen et al. 2013). ● significantly reduced the risk of depredation by **bottlenose dolphins** in a trammel net fishery and increased target catch (Gazo et al. 2008).	20-160 kHz pingers with 145 dB sound level. PROS: ● Pingers were more effective at 'quiet' sites, ● No evidence of habituation by harbour porpoises was observed, and ● Pingers did not affect target and non-target fish size and catch (Hardy et al. 2012). ● Pingers spaced at 455 m apart resulted in 100% reduction in harbour porpoise bycatch compared to 78% reduction in bycatch when pingers 585 m apart (Larsen et al. 2013). CONS: ● There were too few encounters with bottlenose dolphins to determine whether pingers reduced bycatch or to determine how long recolonization to an area would take (Hardy and Tregenza 2010). ● Need to further investigate potential problem of "habituation" - concern that, over time, pingers may alert bottlenose dolphins to nets and may potentially increase depredation rates (Gazo et al. 2008). ● Concern about bottlenose dolphins potentially being excluded from parts of their range if there is widespread use of pingers. Recommended regulation of the use of pingers to reduce negative impacts on dolphin populations in the Mediterranean (Gazo et al. 2008).		To achieve 100% reduction in harbour porpoise bycatch, Aquamark 100 pingers were spaced 455 m apart, although current manufacturer recommendations are for maximum spacing of 200 m (Larsen et al. 2013).

Mitigation measure	Scientific evidence for effectiveness in gillnet fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
		● Evidence that **finless porpoises** (*Neophocaena* spp.) become habituated to pingers after a few months (Amano et al. 2017). ● Limitations with battery - battery life shorter than claimed by manufacturers; unable to use once battery dies as sealed units; high cost per year due to high initial cost and need for replacement devices (Crosby et al. 2013).		
	AQUAmark® 210: In an artisanal gillnet fishery, pingers AQUAmark® 210 significantly reduced **bottlenose dolphin** depredation of bottom-set gillnets, while Dukane Netmark™ 1000 and SaveWave® Dolphinsaver High-impact did not have significant results (Brotons et al. 2008).	5-160 kHz pingers with 150 dB source level.		
	LU-1 prototype pinger: ● significantly reduced **harbour porpoise** bycatch in Danish bottom-set gillnet fisheries (Larsen and Eigaard 2014).	Designed by Loughborough University, England - emitted 8 different signals from 40-120 kHz at random intervals for 300 ms with 145 dB source level. Mandatory use of pingers introduced in 2001 for Danish gillnet fisheries with high porpoise bycatch rates (Larsen and Eigaard 2014).		
	PICE ™ pinger significantly reduced **harbour porpoise** bycatch (Culik et al. 2001).	20-160 kHz at maximum source level of 145 dB ● no difference in herring catches with use of pingers (Culik et al. 2001).		
	SaveWave® (responsive pingers)	To reduce the possibility of habituation, these pingers have a randomized transmission interval and a randomized pulse length with maximum source level of 155 dB. In the North Carolina Spanish mackerel (*Scomberomorus maculatus*) gillnet fishery, **bottlenose dolphin** interactions with gillnets significantly reduce target fish catches. At sea tests showed that: ● pingers did not affect fish catch and dolphins were less likely to interact with gillnets and more likely to echolocate when pingers were present, ● however, these pingers were not sufficiently durable to be deployed in this fishery (Waples et al. 2013).		
	AquaTech UK Continuous Pingers (CP) and Responsive Pingers (RP): ● both CP and RP pingers affected free-ranging **bottlenose dolphin**	20-160 Hz with source level of 165 dB AquaTech interactive pingers (Responsive Pingers) emits signal in response to dolphin echolocations. ● In boat-based trials both CPs and RPs affected **bottlenose dolphin** behaviour. In static trials, detection of dolphin vocalisations significantly lower in the presence of active CP; ● Both types of pingers appear to have the potential to deter **bottlenose dolphins**; ● RPs appeared to be as effective in deterring dolphins as CPs but have the added		

Mitigation measure	Scientific evidence for effectiveness in gillnet fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
	behaviour in Irish fisheries (Leeney et al. 2007).	benefit of not constantly emitting noise into the marine environment and potentially decreasing the likelihood of dolphin habituation to pingers (Leeney et al. 2007).		
	Airmar Acoustic Harassment Device (Airmar Technologies Corporation)	10 kHz with source level of 194 dB. This loud Acoustic Harassment Device (AHD) is deployed in Canadian salmon aquaculture fisheries to reduce interactions with and interference from seals but has been shown to exclude **harbour porpoise** from bays in British Columbia with no porpoises sighted within 200 m of AHD (Olesiuk et al. 2002). Some pinger models may cause harbour porpoises to be temporary displaced from their habitat (Kyhn et al. 2015).		
	Fumunda® F3 (2.7 kHz pinger) **Fumunda® F10** (10 kHz pinger)	132 dB source level. ● Australian snubfin (*Orcaella heinsohni*) and humpback dolphins (*Sousa chinensis*) only subtly changed their movements and behaviour in response to experiments with a Fumunda® F10 pinger. Therefore, it was considered these devices unlikely to be effective in reducing these species bycatch in local small vessel gillnet operations (Berg Soto et al. 2013). ● Acoustic pingers are used to protect marine mammals from bather protection nets (which protect public beaches from sharks) in Australia: - Fumunda® F3 (2.7 kHz) for humpback whales (*Megaptera novaeangliae*) and Fumunda® F10 (10 kHz) for dolphins; - Dugongs detected F3 pinger 90 m from net and F10 pinger 130 m from net, dolphins detected F3 pinger 45 m from net and only detected F10 pinger <40 m from net, humpback whales detected F3 pinger 90 m from net and F10 pinger up to 130 m from net; - pinger output varied with individual pingers and with direction (Erbe and McPherson 2012). ● Fumunda® pingers are some of the quietest commercially available pingers and so, if they are shown to be effective in deterring marine mammals, they may contribute less to ambient noise pollution (Erbe and McPherson 2012). ● Fumunda® F3 pingers did not appear to be effective in deterring bottlenose dolphins from bather protection nets in South Africa as most bottlenose dolphins caught in nets were near pingers. However, controlled experiments to test efficacy are lacking due to the operational requirements of pinger deployment (Erbe et al. 2016).	Erbe and McPherson (2012) only estimated marine mammal detection of sound and did not estimate behavioural responses of individuals - requires further testing to determine whether the sound emitted by these pingers is sufficient to change behaviour, - as only 3 pingers per type were tested, undertaking testing with larger samples sizes would be useful.	3 - 4 pingers per shark net (200 m long) is considered to be adequate in protecting marine mammals from entanglement in shark nets (Erbe and McPherson 2012).
	Fishtek Banana Pinger (developed by Fishtek Ltd, Cornwall, UK) significantly reduced interactions (i.e. detections of echolocation) of **harbour porpoises** and **common** and	Banana Pinger (50-120 kHz with source level of 145 dB) is contained in elastomer housing that stays attached to net - pinger can be easily removed to change the alkaline C-cell battery with battery replacement required about once per year. This contrasts with some other commercial pingers (e.g. AQUAmark®100) which are sealed and need to be replaced once the battery dies. ● low cost pinger that could be used by inshore fleets; ● no decrease in the effect of pingers on **harbour porpoises** over 8 month study period - evidence that habituation of porpoises to these pingers is unlikely to be a	Undertake trials of Banana Pingers on wider range of vessels and in other fishing areas (Crosby et al. 2013).	

Mitigation measure	Scientific evidence for effectiveness in gillnet fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
	bottlenose dolphins (Crosby et al. 2013).	significant issue (Crosby et al. 2013).		
Acoustically reflective nets	**Chemically modified nets**: Barium sulphate nets reduced harbour porpoise bycatch (Trippel et al. 2003; Trippel et al. 2008). Barium sulphate net was approx. 3 times more reflective than standard nylon net and there was significantly less porpoise bycatch in the metal oxide nets (Trippel et al. 2003). Harbour porpoise bycatch was significantly lower in high-density iron oxide gillnets compared to control nets (Larsen et al. 2007). Controlled testing using cetaceans in captivity indicated that bottlenose dolphins may be able to detect metal oxide nets in time to avoid entanglement, but harbour porpoise may not (Mooney et al. 2004).	The monofilament fibres of nets contain a metal compound (Larsen et al. 2007), such as barium sulphate or iron oxide, which may increase the acoustic reflectivity of the net but may also increase net stiffness (Trippel et al. 2003). PROS: ● Barium sulphate nets did not affect target catch (Trippel et al. 2003; Trippel et al. 2008). CONS: ● Barium sulphate nets had higher bycatch of harbour porpoise and seals (common *Phoca vitulina* and grey) compared to traditional gillnets (Northridge et al. 2003). ● There was no significant difference in Franciscana dolphin bycatch between barium sulphate nets and standard nets (Bordino et al. 2013). ● Using Porpoise Echolocation Detectors, there was no difference in echolocation rates between metal oxide and control nets. Concluded that the reduction in bycatch recorded in previously studies is probably unrelated to the acoustic reflectiveness of nets (Cox and Read 2004). ● It is unclear what the mechanism is for the reduced bycatch in chemically modified nets with Trippel et al. (2003) reporting a significant increase in the acoustic reflectivity in barium sulphate nets and Larsen et al. (2007) reporting no difference in reflectivity between iron oxide nets and standard nets. ● Iron oxide nets were much stiffer than standard nets and resulted in much lower target fish catch rates. There was no significant difference in the acoustic target strength between an iron oxide net and control net, and both nets behaved similarly in flume tank tests. Therefore, the reduction of harbour porpoise bycatch in iron oxide nets may be due to mechanical properties rather than the net having any acoustic properties (Larsen et al. 2007). ● Chemically enhanced nets are expensive (Mooney et al. 2004).	Further work required to explain the apparently contradictory results of trials and to further understand the mechanism of why some metal oxide nets had reduced cetacean bycatch (Northridge et al. 2017).	Inconclusive evidence - no recommendations for this measure.
	Passive acoustic additions to nets - metallic bead chains or air-filled plastic tubing	● no evidence that net modifications that included either metallic bead chain or air-filled plastic tubing reduced dolphin bycatch; ● reported higher target catch in unmodified nets (Hembree and Harwood 1987).		
Visually detectable nets	**Addition of net illumination** or **visually contrasting panels**	In a northern Peruvian small-scale fishery, light-emitting diodes (LEDs) on the floatlines of bottom-set gillnets reduced green turtle (*Chelonia mydas*) bycatch by 64% and there was no significant impact on target catch rates (Ortiz et al. 2016). Contrasting patterned panels in gillnets could potentially alert pinnipeds to the presence of a gillnet, although this is yet to be tested. Potential that panels may elicit curiosity rather than aversion in pinnipeds and may negatively impact target catch rates (Martin and Crawford 2015).	Requires research to determine whether net illumination or net panels could reduce cetacean and/or pinniped bycatch.	Currently no recommendations for this measure.

Mitigation measure	Scientific evidence for effectiveness in gillnet fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
Net material strength	Thin twine nets: Significantly less grey seal, common seal and harbour porpoise bycatch in nets made with thin twine monofilament compared to thick monofilament (Northridge et al. 2003).	Seals and porpoises could possibly break free more easily from thin twine nets. Observer data suggested harbour porpoise bycatch higher in nets made from multi-monofilament compared to monofilament, although a paired experiment showed no significant difference in bycatch rates between the two types of nets (Northridge et al. 2003).	Requires research on post-escape impacts on marine mammals, e.g. information on the numbers of animals that retain portions of nets following escape.	Currently no recommendations for this measure.
Weak links		Eastern U.S. fisheries are required to insert weak links in gillnets to allow **large whales** to break free when entangled (NOAA 2018). Various weak link configurations have been trialled to test the operational performance of the modified 'weak link' nets although this work did not include assessment of the efficacy of weak links in reducing cetacean entanglements (Salvador et al. 2006).		Currently no recommendations for this measure.
'Buoyless' nets	Buoyless bottom-set gillnets had significantly less sea turtle bycatch while maintaining target catch rates (Peckham et al. 2016).	Standard nets targeting a range of groundfish species in northwest Mexico bottom-set gillnets have about 1 buoy every 1.7 m of float line, whereas experimental "buoyless" nets had only 1 buoy every 8.5 m. Reducing the buoys from float lines of conventional nets was assessed to determine the effects of this net modification on both turtle bycatch and target catch rates. The buoyless net design is likely to decrease the vertical net profile of bottom-set gillnets. ● potential to also reduce bycatch of cetaceans and pinnipeds, although this is yet to be evaluated; ● although target catch volume was not greatly different in modified nets, the market value of buoyless net catch was significantly lower; ● modified nets may have increased net tangling rates; ● water depth and water visibility prevented underwater observation of functioning of buoyless nets to fully understand the mechanism of decreased bycatch (Peckham et al. 2016).	Requires research to determine whether buoyless bottom set gillnets could reduce cetacean and/or pinniped bycatch.	Currently no recommendations for this measure.

Appendix 2E: Technical mitigation measures developed to reduce marine mammal bycatch in pot and trap fishing operations.

General description of pot and trap fishing and bycatch: Pots and traps are enclosures that trap or entangle fish, crustaceans or molluscs by enticing them inside, either using bait or because the enclosure appears to provide a refuge. Depending on the target species, most pots and traps have one or more entrances on the top or sides, are set on the seabed or riverbed, and have a haul-in line, surface float or dan buoy to mark their position. In this chapter, a pot refers to a small baited enclosure (e.g. lobster pot) while a trap refers to a larger un-baited structure (e.g. Newfoundland cod trap).

Marine mammal bycatch mainly occurs when pinnipeds and small cetaceans become trapped after entering pots/traps to feed on captured target species (Campbell et al. 2008; Konigson et al. 2015) or large baleen whales become entangled in the associated lines of a fishery (Benjamins et al. 2012; Groom and Coughran 2012; Knowlton et al. 2012; Lloyd and Ross 2015; Vanderlaan et al. 2011). Although some large whales may break free, any gear remaining entangled around an animal can impact foraging and reproduction, increase drag, and cause infection, haemorrhaging and/or severe tissue damage which may result in mortality (Kraus et al. 2016; Moore and van der Hoop 2012). Significant numbers of threatened Australian sea lions (*Neophoca cinerea*), particularly juveniles and pups, have drowned after becoming trapped in rock lobster pots (Campbell et al. 2008) and substantial bycatch of harbour and grey seals has been recorded in a Baltic Sea cod pot fishery (Konigson et al. 2015; Stavenow et al. 2016).

Mitigation measure	Scientific evidence for effectiveness in pot/trap fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
Reducing entrapment mortality in pots/traps				
Pot/trap excluder devices, e.g. trap guards and pot spikes	**Bungee cord trap guards in crab pots** ● reduced interactions between bottlenose dolphins and a Florida crab pot fishery (Noke and Odell 2002).	NB: reducing bottlenose dolphin interactions primarily about reducing depredation of bait fish in pots		No current recommendations or standards. Need to be assessed and refined for each fishery issue.
	Sea Lion Excluder Devices in rock lobster pots ● SLEDs (either upright steel rod attached to pot base and rising up to centre of pot opening, or a steel bar batten across pot opening) virtually eliminated vulnerable-sized Australian sea lions from entering rock lobster pots (Campbell et al. 2008). ● Field trials in July 2017 showed Australian fur seal (*Arctocephalus pusillus*) pups aged approximately 7 months old were not able to pass through a ring with 150 mm opening. Concluded that the minimum diameter to prevent seal pups entering is	SLED designs trialled in commercial Western Australian western rock lobster (*Panulirus cygnus*) pots to stop sea lions entering the pots included: A) steel rod pointing up from pot base towards pot opening (20 mm below pot collar) creating max. SLED-neck gap of 132 mm, B) batten across pot opening also with neck gap of 132 mm. ● In 0-20 m water, 'SLED A' had no impact on target catch although 'SLED B' reduced target catch by 14%; ● In deeper water, both designs caught significantly fewer legal-sized lobsters than standard pots. As majority of sea lion pup and juveniles dives were <20 m, a mandatory SLED zone in west coast Australian rock lobster fishery depths <20 m was established in 2006/07 (Campbell et al. 2008). In South Australian waters, sea lion pups and juveniles dive almost 3 times deeper than those studied in west coast Western Australia (Fowler *et al.* 2007). Indication from trials that appropriately configured spikes could reduce the likelihood of juvenile sea lions entering southern rock lobster (*Jasus edwardsii*) pots in southern Australian fisheries although there was low statistical power in preliminary experimental testing of 3 different designs: ● Steel bar (metal bar secured across pot neck) - stopped sea lions entering pot but reduced rock lobster catch by 18%;	In South Australian waters: - SLED spike designs require further testing to ensure adequate statistical power; - testing of a spike extended to the base of the pot-collar and further reduction of minimum pot opening could prevent smallest pups from fully entering pots (Goldsworthy et al. 2010).	Two SLED designs (either with upright steel rod or batten across the pot-opening) that ensure an pot opening ≤ 132 mm are mandatory for west coast Western Australian rock lobster pots in waters < 20 m (Department of Fisheries 2012; DSEWPaC 2013). In the South Australian Northern Zone Rock Lobster Fishery, a spike SLED has been mandatory in commercial lobster pots in < 100 m waters

Mitigation measure	Scientific evidence for effectiveness in pot/trap fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
	135 mm side length for 'rectangular or square' opening SLEDs and 150 mm diameter for 'circular opening' SLEDs (Mackay and Goldsworthy 2017).	● T-bar (upright bar welded to base of pot with cross piece attached to top) - reduction in sea lions entering but still some predation of rock lobsters and reduced catch rate by 2%; ● Double neck (increased neck length of pot) - sea lions still able to enter pots (Goldsworthy et al. 2010). Although the currently mandated spike SLED design reduced interaction between Australian sea lions and lobster pots, there are operational concerns about their use in commercial fishing conditions. Therefore, further experimental testing of the following two industry designed SLEDS in South Australia's southern rock lobster fishery (NB: standard lobster pot has 270 mm diameter neck opening): 1) "squeezy neck" – pot with modified collar with 135 mm x 270 mm metal rectangle opening; 2) "box-pot" – box-shaped pot with three 150 mm x 245 mm openings. Testing was undertaken using Australian sea lion pups as well as long-nosed fur seal pups. ● The "squeezy neck" design with a 135 mm narrowest opening should prevent entry of small sea lions; ● A 150 mm diameter ring opening excluded all Australian sea lion and long-nosed fur seal pups tested; ● Results were inconclusive on whether a box-plot SLED with minimum side length of 150 mm would fully exclude seal pups; ● Further work is planned to assess lobster catch rates in pots with different SLED designs (Mackay and Goldsworthy 2017).		since November 2013 (Mackay and Goldsworthy 2017).
	Seal Excluder Device (SED) in cod pots ● bycatch of grey and harbour seals was reduced to zero when SEDs were used on Atlantic cod pots (Konigson et al. 2015).	Mainly used two-chambered single-entrance floating cod pots. SEDs design had vertically mounted metal frames secured with nylon at narrow end of pot's entrance to prevent juvenile grey and harbour seals from entering pots. Five different SED designs tested - different shapes, thicknesses or inner circumferences (either 54 cm, 56 cm, 64 cm or 70 cm). ● SEDs with symmetrical oval shaped entrances as well as larger rectangle shaped entrances increased the target (cod) catch rate of pots (Konigson et al. 2015).		No current recommendations or standards. Need to be assessed and refined for each fishery bycatch issue.
	Wire grid guards in the funnel of salmon trap-nets ● wire grids along with fish bags made of seal-safe Dyneema® netting minimized seal-induced damage and reduced the incidental mortality of ringed seals (*Phoca hispida botnica*) and grey seals caught in coastal salmon trap-nets (Hemmingsson et al. 2008; Suuronen et al. 2006).	Baltic Sea coastal trap-net fisheries utilise floating, bottom-anchored trap-nets to catch salmon. ● refinements of wire grid design (e.g. grid spacing) was critical for the seal-safe trap-net to be effective, ● target catch remained the same or increased in modified traps (Suuronen et al. 2006).	Need to develop designs that keep seals away from trap-net wings and middle chambers (Suuronen et al. 2006).	No current recommendations or standards. Need to be assessed and refined for each fishery issue.

Mitigation measure	Scientific evidence for effectiveness in pot/trap fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
	Decreasing trap opening size ● decreasing opening size of crab fyke traps was likely to reduce sea otter (*Enhydra lutris*) bycatch in a Californian crab fishery (Hatfield et al. 2011).	Tested sea otter interactions to finfish traps, lobster traps and crab fyke traps in captive trials - some animals became trapped after entering circular and rectangular fyke openings. ● reducing fyke openings from 10.2 x 22.9 cm to 7.6 x 22.9 cm would exclude most free-living (i.e. weaned from mothers) otters, ● the reduced opening size was unlikely to impact target crab capture, ● currently no observer program in California commercial Dungeness crab (*Metacarcinus magister*) fishery and it is possible that sea otter bycatch is undetected (Hatfield et al. 2011).	Requires testing in a commercial fishery.	No current recommendations or standards. Need to be assessed and refined for each fishery issue.
"Seal socks"	"Seal sock" was effective in reducing bycatch of ringed seals in coastal fyke nets in the Baltic Sea, although was not as effective for grey seals (Oksanen et al. 2015).	Baltic Sea coastal fixed gear fishery uses fyke nets (i.e. large, bottom-anchored leader net with wings which guide fish through a middle chamber and into a fish chamber which is usually set 2 m deep from surface) to target *Coregonus* spp., Salmonidae and *Salmo* spp. Baltic grey seals (*Halichoerus grypus baltica)* and ringed seals (*Phoca hispida botnica*) enter fyke net chambers and can become trapped and drown. "Seal sock": cylindrical net attached to fish chamber section of coastal (<2 m deep) fyke nets and allows trapped seals access to surface to breathe while still inside fishing gear. Notes: ● bycatch mainly affects juvenile seals; ● all seals (n=77) died in fyke nets without a "seal sock" whereas 70% of ringed seals (n=40) and 11% of grey seals (n=18) survived in nets with a "seal sock"; ● ringed seal pups less likely to survive even with a "sea sock"; ● behavioural differences between adult grey seals and adult ringed seals may explain difference in survival rates with "seal sock" - ringed seals typically access breathing holes in sea ice and may be more capable of swimming upwards in a narrow funnel; ● adult grey seals mainly caught in middle chamber and could not pass through narrow passage between chambers and so could not access "seal sock" (Oksanen et al. 2015).		No current recommendations or standards. Need to be assessed and refined for each fishery issue. In this study, "seal socks" effective for traps set <2 m deep.
Reducing interactions with and mortality from ropes/lines in water column				
Acoustic Deterrent Devices (Pingers)	**Lien model** ● pingers appeared to reduce humpback whale collisions with cod traps in Canadian coastal cod trap fishery (Lien et al. 1992).	Louder devices which produce sounds centred on 4 kHz (peak frequency 135 dB re 1~Pa at 1m) Pros: ● traps with pingers caught more fish than traps without pingers, ● good cooperation and support from fishers during trials. Cons: ● whales response to pingers may change over time, ● potential noise pollution in marine environment (Lien et al. 1992).	Research on whether the effect of pingers deterring whales declines over time. Research on the impact of widespread pinger use on other marine species.	May be effective for some species at certain times of year. Standards yet to be recommended.
	Acoustic deterrents In a behavioural response study, southward migrating humpback whales (mainly	Northward migrating humpback whales (mainly adults including near-term pregnant females) showed no detectable response (direction, dive duration and speed) to different pingers: - 3 kHz commercial whale alarm, 400 ms tone repeated every 5 s (Harcourt *et al.*	Research the effect of different types of pingers on migrating whales (Pirotta et al. 2016).	

Mitigation measure	Scientific evidence for effectiveness in pot/trap fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
	females with newborn calves) along Australia's east coast responded to 2 kHz swept tone stimuli with 1.5 s tone repeated every 8 s (Dunlop et al. 2013).	2014); - 5.3 kHz tone (5 s emission interval and 400 ms emission duration), and - 2-2.1 kHz swept tone (8 s emission interval and 1.5 s emission duration) (Pirotta et al. 2016). Preliminary trials of pinger arrays (Future Ocean F3 with 135 dB) in Western Australia showed that, overall, no statistical difference in southward migrating whale pod behaviour (change in direction) due to presence or absence of pinger activity. However, in one analysis there was weak evidence that whale pods may change direction when pingers were active but this testing requires an increased sample size to improve the statistical rigour of the results (How et al. 2015). On the Oregon coast, there were indications that grey whales (*Eschrichtius robustus*) could be deterred from an area using an acoustic device (1-3 kHz with source level of 170 dB), although due to poor weather and equipment issues, interaction rates were low resulting in low power in statistical analysis (Lagerquist et al. 2012). CAVEATS: ● whale responsiveness to pingers may differ depending on migration direction and social category, ● whales may be accustomed to wide range of anthropogenic noises when travelling along populated coastlines and so may not be readily deterred by pingers (Pirotta et al. 2016). ● single alarms attached to pot/trap lines unlikely to deter humpback whales from approaching potential hazard areas (Harcourt et al. 2014).		
Negatively buoyant or sinking groundlines		Groundlines, which connect pots/traps to each other in strings, often have floating sections which may pose a risk to passing whales. **Negatively buoyant or 'sinking' groundlines** aim to lie closer to the ocean bottom and reduce entanglement risk to whales compared to groundlines constructed of rope that floats (Brillant and Trippel 2010). Broad-based sinking groundlines have been mandatory in U.S. pot/trap fisheries since April 2009 (Knowlton et al. 2012). CAVEATS: ● Technique has been unpopular as sinking lines may be more like to get caught or abraded on rocky ocean floor than if floating in the water column. ● Difficult to ascertain effectiveness as lack of understanding of dynamics of whale entanglement including where in the water column entanglements are likely to occur. ● Technique often implemented as part of a range of mitigation measures, so difficult to determine effectiveness of each mitigation component (Laverick et al. 2017). However, the range of measures implemented in U.S. fisheries over a number of years has not reduced entanglement rates of northern right whales (*Eubalaena glacialis*) to sustainable levels (Knowlton et al. 2012). ● Negatively buoyant groundlines are considered unlikely to reduce entanglement rates for northern right whales (Brillant and Trippel 2010).	Further research is required to determine the effectiveness of sinking groundlines.	In north-eastern U.S., a number of regulations apply to pot or creel fisheries to protect large whale species, particularly northern right whales, including the prohibition of floating groundlines; groundlines must be made of sinking line (Northridge et al. 2010). However, there is a lack of evidence to recommend this as a mitigation measure.

Mitigation measure	Scientific evidence for effectiveness in pot/trap fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
		Conversely, entanglements of minke whales (*Balaenoptera acutorostrata*) mainly occur around the mouth and involve groundlines and branchlines rather than buoylines (Song et al. 2010).		
Rope-less systems	**Galvanic time releases** and **acoustic releases of buoy marker systems**	Devices that secure vertical fishing lines to the bottom until they are released for hauling. These aim to reduce the time that vertical lines are in the water column and, therefore, reduce cetacean entanglement risk. CAVEATS: ● issues with storage of some prototypes on board vessels; ● design needs to ensure storage of adequate rope length for relevant fishery; ● some system designs likely to have high initial costs (How et al. 2015; Partan and Ball 2016); ● Western Australia lobster fishery trials involved a small sample of remote-released pots due to low uptake as fishers were concerned about gear loss and high costs (How et al. 2015; Laverick et al. 2017); ● limited assessment of impact on target catch rates; ● no publications verify that these devices are effective in reducing bycatch and practical and cost-effective for implementing in operational fisheries (Laverick et al. 2017). Range of preliminary studies: 1) In Gulf of Maine and southern New England lobster pot fishery, acoustic release system for offshore lobster buoy lines developed for reducing northern right whale entanglements with vertical lines - gear successfully developed and at-sea testing confirmed feasibility of gear (DeAlteris 1999). 2) Initial trials of a Buoyless Lobster Trap (BLT) system with electronics, acoustic release trigger and winding mechanism components. Rotary solenoid released a buoy and line from the gear on the ocean floor up to surface. The acoustic trigger mechanism could be used on existing lobster traps. No testing on impact on bycatch species or on target catch (Turner et al. 1999). 3) Initial trials of various vertical line release options were undertaken, e.g. acoustic release unit for lobster pot buoy line; galvanic timed releases of vertical lines (Salvador et al. 2006). 4) Prototypes of 'on-call' buoy systems consisting of a flotation line spool capable of holding up to 900 m of ½ inch line secured to a lobster trawl by a mechanical release that can be triggered for hauling either using digital timer or acoustic transponder. Preliminary investigation on passive acoustic detectability of rope-less gear showed echo-sounder detection of rope-less endline spools in 300 m is likely feasible (Partan and Ball 2016). 5) Anode release and acoustic release system trialled by fishers in Western Australia (How et al. 2015). Devices deployed in operational lobster fishery in New South Wales (Australia): Release mechanism in New South Wales lobster fishery device based on vaporization of a nickel-chromium wire using very high current electric current (http://www.desertstar.com/page/arc-1xd). If acoustic release (AR) fails, a back-up galvanic time release (GTR) is triggered after a given time period to release gear. Conclusions based on 6 months of commercial fishing at 100-120 m depths using these devices:	Design refinement and research required to develop reliable deployment systems as well as devices that can contain enough rope for fisheries that operate in deep water. Designs need to be tailored to operational requirements and conditions of different fishing and operational testing of devices is required (Partan and Ball 2016; Salvador et al. 2006).	Acoustic release mechanisms have been used for a number of years in New South Wales (Australia) Rock Lobster Fishery (Liggins 2016).

Mitigation measure	Scientific evidence for effectiveness in pot/trap fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
		- devices maintained their integrity at depth and did not leak, - battery voltage longevity and power consumption was acceptable, - detection range is practical using GPS, - electronics and release mechanism remained functional, - loss rate of traps was less with AR compared to GTR, - lobster catch rates were greater with AR than GTR traps (Hodge 2015).		
Rope colour	● Surface feeding northern right whales had a significantly earlier response to red and orange ropes compared to black and green ropes in trials using simulated lobster buoy ropes (Kraus et al. 2014; Kraus and Hagbloom 2016). ● Trials of different coloured experimental ropes and buoys, simulating those used in crab and whelk fishing gear, showed that minke whales (*Balaenoptera acutorostrata*) decreased swimming speed and changed direction when approaching black and white ropes (Kot et al. 2012).	Different whale species appear to have different responses to red, orange or yellow ropes: ● **Northern right whales** are visually tuned to perceive red and orange as high contrast 'black' against ambient blue/green oceanic background. Green and black ropes are commonly used in Maine lobster fisheries. Experimental testing of green, black, red and orange 'simulated' (PVC) ropes showed whales had greater responses to red and orange ropes (Kraus et al. 2014). ● Orange ropes and yellow mainlines were significantly over-represented in assessments of **humpback whale** entanglements in Australia. However, it is unknown whether humpback whales are more likely to visually detect orange and yellow colours and are attracted to them, or whether they are less likely to detect those colours (How et al. 2015). ● Kot et al. (2012) trialled yellow, orange, green, blue, white, and black ropes to test if **minke whales** behaved differently to different colours. In low light, the passive acoustic abilities of whales may assist rope detection rather than visual detection.	Further research required to test right whale detectability of different rope colours in a range of conditions e.g. varying distances and underwater visibilities (Kraus et al. 2014). Research also required on the effect of rope colour for other whale species and also on determining how the detectability of ropes may influence the likelihood of whale interactions.	No recommended standards although Kot et al. (2012) recommend that, to reduce minke whale entanglement, gear that contrasts best with the surrounding fishing areas should be used.
Rope strength	**Weak rope** Large whales may be more likely to break free if gear is suitably modified.	Rope used in modern fishing is usually made from polypropylene which is much stronger and more durable than the hemp and sisal ropes previous used. A study of 132 ropes removed from 70 entangled (both live and dead) baleen whales investigated the impact of rope polymer type, breaking strength and diameter on injury severity. Considered that Reduced Breaking Strength (RBS) ropes with breaking strengths of ≤ 7.56 kN could reduce the number of life-threatening whale entanglements by at least 72% (Knowlton et al. 2016). PROS: ● RBS with breaking strengths of ~7.56 kN likely to still tolerate forces of fishing operation; ● Compared to weak links which only break at a certain point, RBS ropes are designed to break wherever the whale contacts with the rope. CONS: ● RBS ropes would not reduce interactions between whales and gear; ● RBS ropes may not prevent severe entanglements of young animals or calves (Knowlton et al. 2016).	Need to ensure RBS ropes are feasible for different fishing conditions. Research is required to determine the effectiveness of RBS ropes in an operational fishery. Requires research on post-release injuries and mortality.	Maximum breaking strength of vertical line in most southeast U.S. waters has been set at 9.8 kN, with 6.7 kN in Florida state waters (Leaper and Calderan 2018).

Mitigation measure	Scientific evidence for effectiveness in pot/trap fisheries	Caveats/Notes	Research needs	Minimum standards/ recommendations
Weak links and line cutters	**Weak links**	Weak links connect the vertical line to the buoy system and are designed to break under particular level of pressure (Knowlton et al. 2012). Weak links are required for U.S. lobster fisheries (Knowlton et al. 2012; Van der Hoop et al. 2013). Current specifications: weak link with a max. breaking strength of 272.4 kg on all buoys for lobster trap/pot gear in inshore waters, and weak links with max. breaking strength of 680.4 kg on all buoys in offshore areas (Laverick et al. 2017; NMFS 2015). CONCERNS: ● weak links do not appear to have reduced the incidence or severity of large whale entanglements (Knowlton et al. 2012; Knowlton et al. 2016; Pace et al. 2014; Van der Hoop et al. 2013). ● as the buoy is rarely involved in the actual entanglement, break-away links located so that the buoy is released will not necessarily release the whale from entangled bits of rope (Moore 2009). ● disentanglements are more logistically feasible and successful when the whale is anchored and easily located compared to whales that are released but continue to have gear entangled around them (How et al. 2015; Laverick et al. 2017).	Requires further research to determine if weak links decrease the entanglement risk for whales. Requires research on post-release injuries and mortality.	Currently required for some U.S. fisheries. However, concern about the efficacy of weak links in reducing entanglements and whether whales retain sections of gear. Not recommended as a mitigation measure in Western Australian lobster fishery due to concerns that released whales may retain sections of rope and it may be more difficult to disentangle them once they are free-swimming (How et al. 2015).
	Time Tension Line Cutter (TTLC)	Device designed to address large whale entanglements with buoy lines. A buoy line is released from anchored gear when a cutting blade is triggered by a specific line load being exceeded for a predetermined length of time. ● Preliminary trials indicated that release mechanism performed as programmed but the loads experienced in the trials were less than would be present in a commercial fishing operations and more work is required to address the logistic issue of hauling gear equipped with a TTLC (Salvador et al. 2008).		
Stiff ropes		Stiff ropes are designed to be stiff in water column and loose on boat deck - increasing rope stiffness or tension could reduce entanglement risk as less flexible rope may allow animals to slide off more easily. ● Consortium for Wildlife Bycatch Reduction (2014) tested whether ropes with greater stiffness or greater tension may reduce entanglements, measured rope tension under different environmental conditions, and studied fisheries that use tense rope to determine how effective it may be to reduce bycatch. No published reporting of this research. ● experimental testing of the interaction between a tense vertical line and a physical model of a northern right whale flipper indicated that stiff ropes may increase the probability of lacerations especially if the first point of contact is the whale flipper (Baldwin et al. 2012).	Requires reporting on research to date.	Not recommended as a mitigation measure.

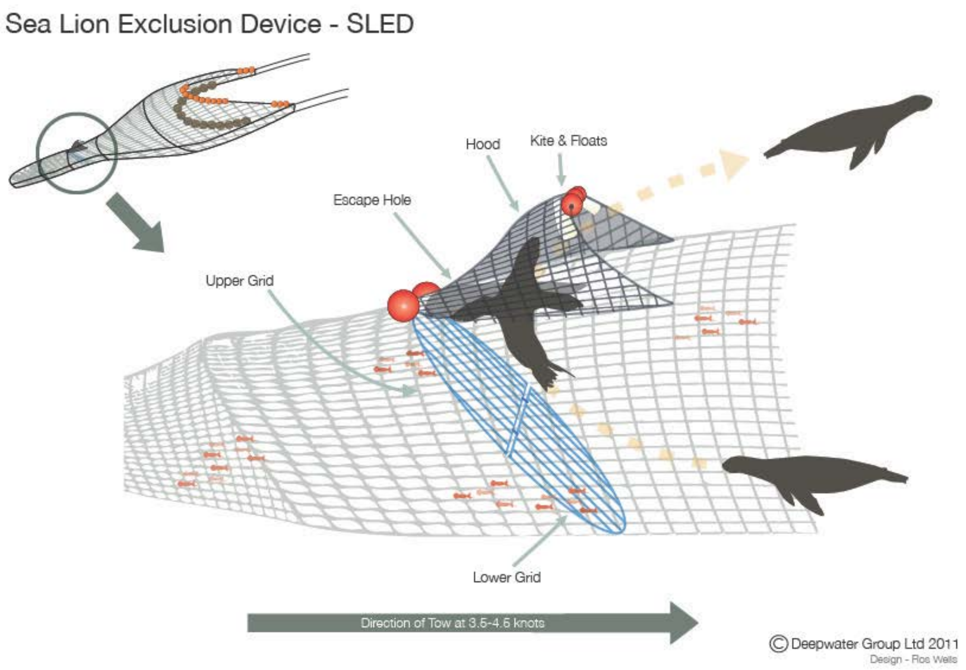

Figure 2A.1: Standard New Zealand Sea Lion Exclusion Device (SLED) used within the Auckland Islands squid trawl fishery. Figure provided by Deepwater Group Ltd., New Zealand and published in Hamilton and Baker (2015a).

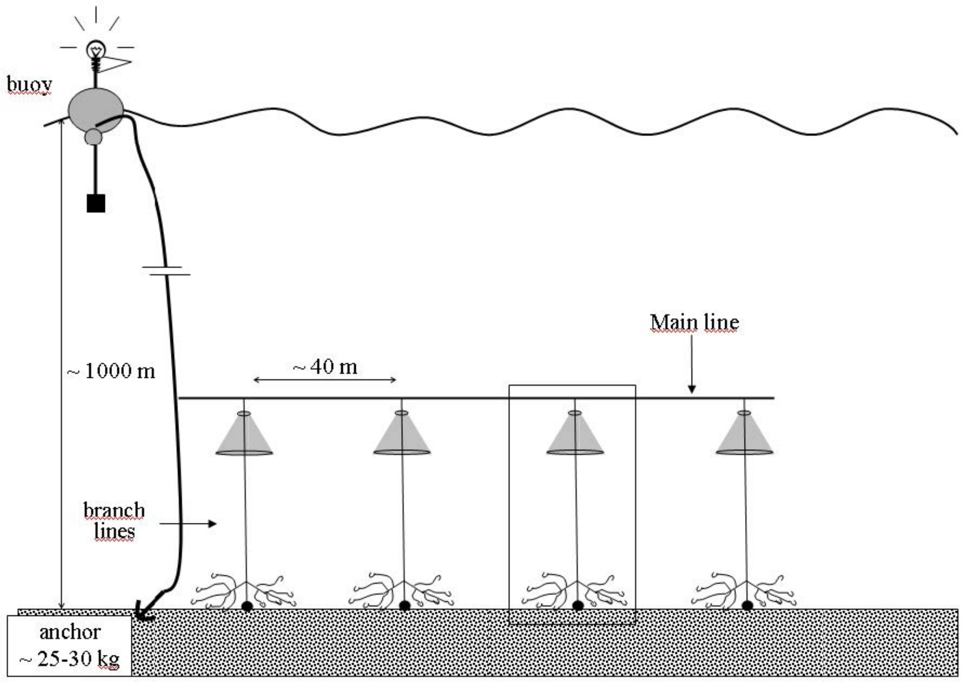

Figure 2C.1: Chilean longline system with main line and secondary branch lines, with multiple hooks and net sleeves or "cachaloteras" which slide down over the hooks during the haul. Figure modified from Moreno et al. (2008).

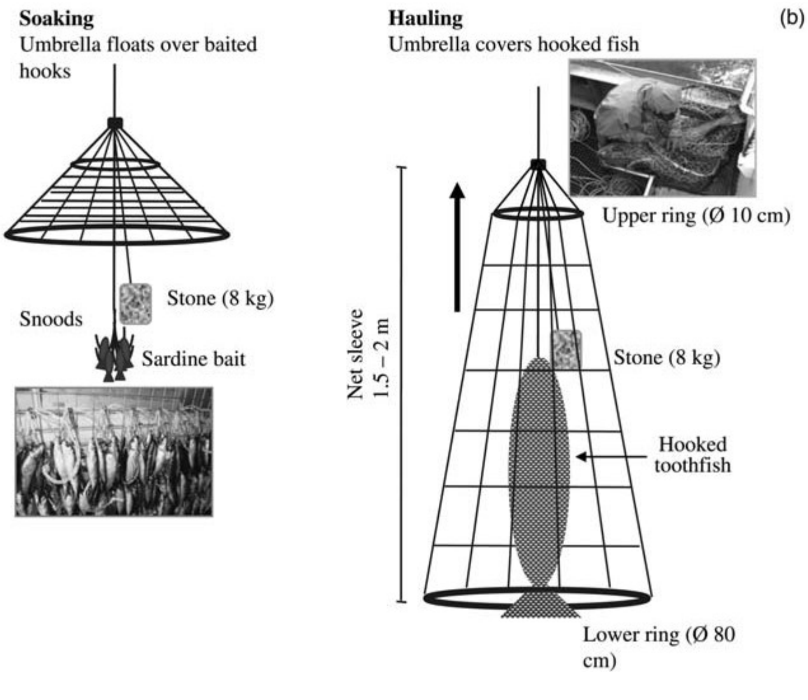

Figure 2C.2: The "umbrella and stone" design and mechanism. Figure modified from Goetz et al. (2011).

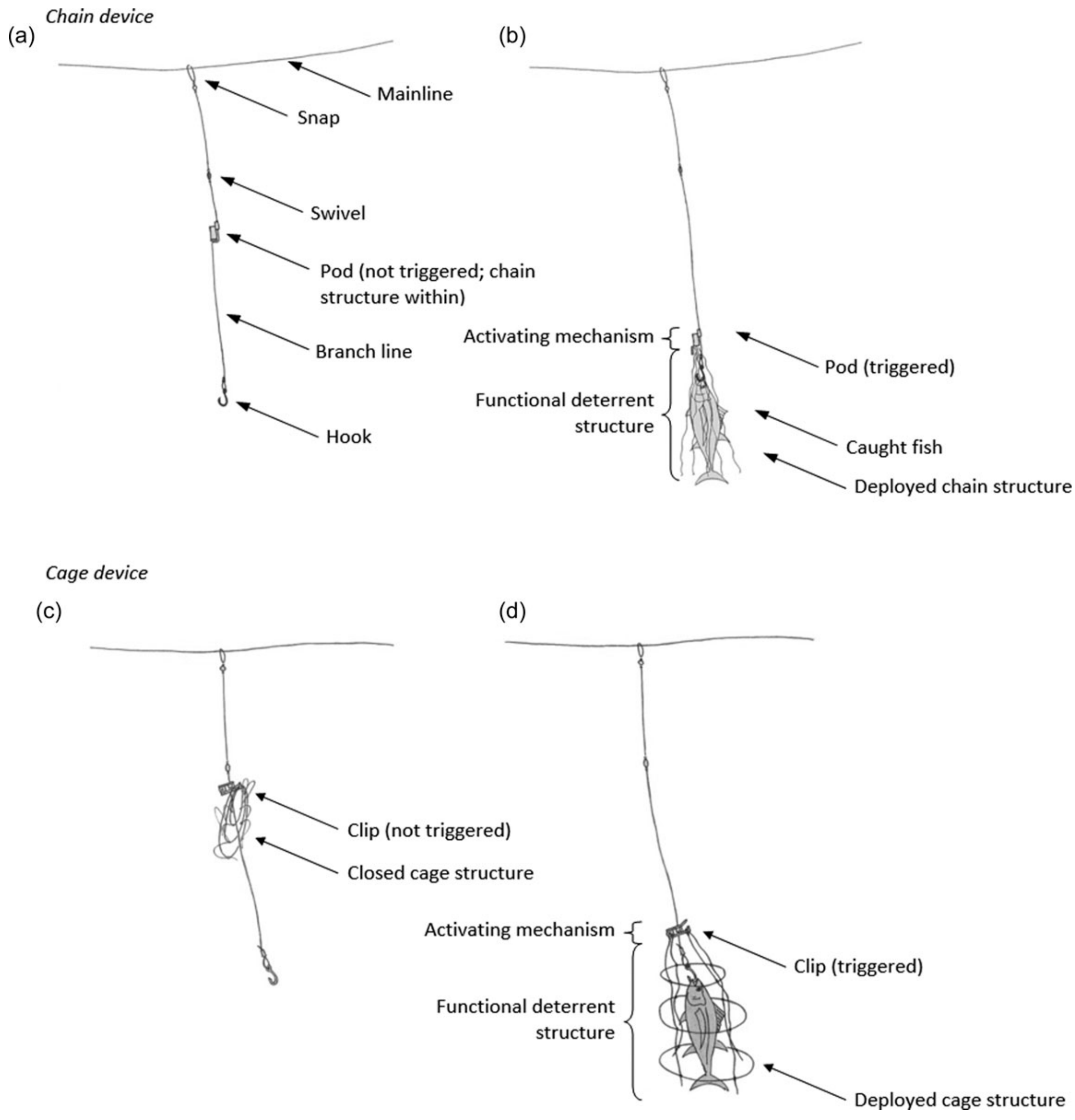

Figure 2C.3: "Chain" and "cage" devices (a and c, not triggered; b and d, triggered) designed to physically deter depredating odontocetes, thus also mitigating the risk of bycatch. Diagram from Hamer et al. (2015).

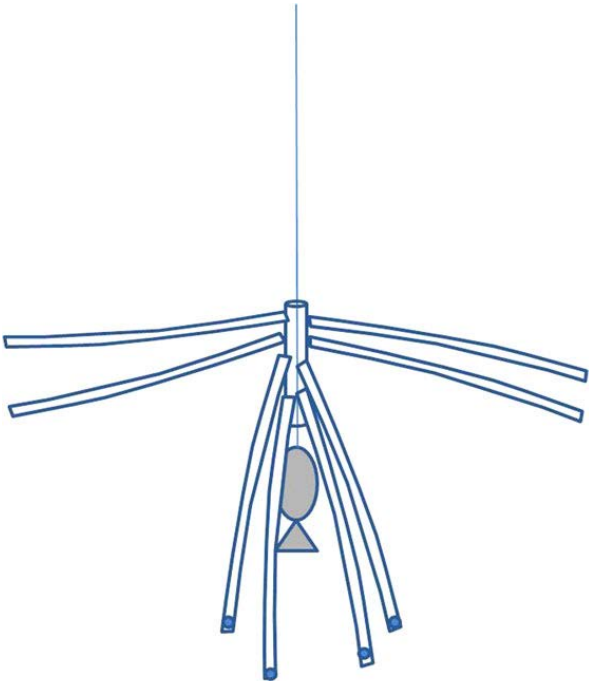

Figure 2C.4: The DEPRED (DEPREDation mitigation device by preventing predator attacks and protecting capture) is a depredation mitigation device made up of eight streamers. Upper ones freely move around the fish (deterrent affect) and lower ones are weighted, covering the fish (protective effect). Figure from Rabearisoa et al. (2015).

Chapter 3 — Review of research and assessments on the efficacy of sea lion exclusion devices in reducing the incidental mortality of New Zealand sea lions *Phocarctos hookeri* in the Auckland Islands squid trawl fishery

Published as:

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Updated and developed in the context of my thesis framework, with additional data and research incorporated – no change to overall findings or conclusions.

3.1 Abstract

New Zealand (NZ) sea lions are incidentally killed in the Auckland Islands squid trawl fishery. Sea lion exclusion devices (SLEDs) that allow animals to escape from the trawl net have received considerable development and assessment attention. Nonetheless, there are claims that some animals could suffer head trauma when colliding with the hard grid that forms part of the SLED and this may compromise post-escape survival, or they may still be dying in nets but their bodies may passively drop out the SLED escape hole. I reviewed published and unpublished research that assessed the effectiveness of SLEDs in reducing the incidental capture (i.e. bycatch) of sea lions, including assessments on the likelihood of post-SLED survival.

The available evidence shows that SLEDs are effective in reducing sea lion bycatch in trawl nets and contribute to reduced rates of observed sea lion mortality in the Auckland Islands squid fishery. Efforts to test SLED efficacy have shown that most NZ sea lions are likely to survive following their escape via a SLED, despite the shortage of verifying video evidence mainly due to poor visibility at fishing depths. Laboratory necropsies of incidentally caught sea lions have been unable to reliably evaluate post-SLED survivability of sea lions due to the effects of on-board handling of carcasses,

including the logistical necessity of freezing them. Some lesions initially considered to be evidence of trauma were subsequently deemed to be artefacts of freezing. Furthermore, there was no clear difference in the trauma assessments between sea lions caught in nets with and without SLEDs. Biomechanical modelling suggested it was unlikely that impact with a SLED would cause fatal brain trauma and the probability of concussion that could result in post-SLED drowning was probably less than 10%. Additionally, there is no evidence that dead sea lions would drop out of top-opening escape holes in an effectively functioning SLED and it is unlikely this contributes substantially to any cryptic or unaccounted mortality. As fisheries bycatch has been reduced to levels that are unlikely to be driving continued decline of the Auckland Islands population of NZ sea lions, future work may be better focussed on alternative research and management areas that may be more effective in addressing and reversing NZ sea lion population decline, such as disease at colonies and management of natural hazards that reduce survival of pups.

3.2 Introduction

The endemic New Zealand (NZ) sea lion/rāpoka (*Phocarctos hookeri*) is listed as Endangered on the IUCN Red List (Chilvers 2015) and as Nationally Vulnerable on the NZ Threat Classification system (Baker et al. 2019). Historically, subsistence hunting followed by commercial sealing greatly reduced both its population and breeding distribution and, although formerly breeding throughout NZ, the species is now largely confined to the NZ sub-Antarctic (Childerhouse and Gales 1998; Childerhouse et al. 2010b). Based on pup production estimates, 68% of all sea lion pups are born at the Auckland Islands, about one third at Campbell Island with the remainder recorded from Stewart Island and the Otago coast, with additional haul-out sites on The Snares as well as Macquarie Island (DoC and MPI 2016) (Figure 3.1). The most recent total population estimate, calculated for the 2014/15 season, was 11,800 sea lions (including pups) (Roberts and Doonan 2016) compared to an estimate of less than 12,000, including pups, in 2006 (Maloney et al. 2009).

Over the last twenty years, commercial trawl fisheries have been implicated in the observed decline of the sea lion population at the Auckland Islands, due to the incidental mortality (hereafter referred to as “bycatch”[1]) of sea lions in trawl nets (Robertson and Chilvers 2011). Annual estimates from all breeding locations in the Auckland Islands showed sea lion pup production, which is the best index of relative overall population size for this species, decreased from a peak of 3,021 pups in

[1] The term “bycatch” is used to describe the accidental capture of sea lions in trawl nets and I have assumed that captures are the equivalent to mortalities.

1997/98 to a low of 1 501 in 2008/09 (Childerhouse et al. 2017; Chilvers et al. 2007; Robertson and Chilvers 2011). However, all eleven annual counts since 2008/09 have been above 1,501 (with highs of 1,940 in 2012/13 and 1,965 pups in 2016/17) and there appears to be a fairly stable trend with the most recent estimate being 1,740 pups in 2019/20 (Melidonis and Childerhouse 2020). Most sea lion bycatch has occurred in the Auckland Islands squid fishery, although bycatch has also been recorded in the Auckland Islands scampi fishery, Auckland Islands non-squid/scampi target fisheries, the Campbell Island southern blue whiting fishery and Stewart-Snares shelf trawl fisheries (Thompson et al. 2013b). Sea lions may become caught when they opportunistically depredate fish from trawl nets, because the probability of fish being hauled nearer to the surface is high and because these events negate the need to undertake energetically expensive dives to search for prey (Hamer et al. 2013). They may choose to dive on a net when it is being set to depredate fish enmeshed from the previous fishing event (known as ‘stickers’) or may use operational cues to dive on the net as it is being hauled to depredate freshly caught fish. Alternatively, sea lions may be caught when they are foraging naturally in the same areas and depths that trawl nets are operating.

The annual Auckland Islands squid fishery targeting *Nototodarus sloanii* is one of NZ's largest and more valuable fisheries, using a combination of bottom and mid-water trawls operating across the shelf at bottom depths of about 150 - 250 m (MAF 2012). Peak activity in the fishery occurs between February and May (MAF 2012), which coincides with part of the lactation period for breeding female sea lions when lactating mothers are regularly and frequently foraging at sea (Chilvers 2008a). Although the foraging areas and depths of sea lions are not completely understood, tracked sea lions have been shown to overlap, in part, with commercial trawl fishing activity in the Auckland Islands squid fishery (Chilvers 2008a; Chilvers 2009).

The incidence of sea lion mortality in the Auckland Islands squid fishery has been monitored by government observers since 1988 (Wilkinson et al. 2003), although observer coverage has varied from less than 10% to 99% (Figure 3.2). The mean estimated level of bycatch peaked in the mid to late 1990s (specifically at 131 in 1995/96 and at 142 in 1996/97; Thompson *et al.* 2013). As a result, sea lion conservation management has focused on this fishery, with a number of research projects commissioned to assess the impacts of sea lion bycatch and the development and assessment of new approaches to reduce those impacts. Management aimed at mitigating sea lion bycatch includes the establishment of a 12 nautical mile marine reserve around the Auckland Islands which excludes all fishing within that range, the imposition of mortality limits that can trigger spatio-temporal closures, and the development and implementation of a ‘Sea Lion Exclusion Device’ or SLED (Figure 3.3; MAF (2012)). The SLED comprises an additional section of netting inserted between the lengthener and codend of the trawl net with an angled two or three part metal grid that is designed to direct sea lions to an escape hole in the top of the net and exclude them from the trawl codend (Abraham 2011; MAF 2012; Middleton and Breen 2011; Roe and Meynier 2012; Wilkinson et al. 2003). A standardised SLED design for the fishery has been widely adopted. This includes grid bar spacing of 23 cm to reduce the probability of smaller sea lions passing through the grid and becoming trapped in the codend, a hood and kite fitted to the top-mounted escape hole, and additional floats incorporated on the top of the hood to ensure the kite and hood operates properly in all conditions and the escape hole remains open during fishing (MPI 2012). Similar exclusion devices have been developed in Australian fisheries to reduce the mortality of fur seals although exclusion devices can differ significantly between fisheries, particularly in terms of the location of the escape hole (Hamer and Goldsworthy 2006; Lyle et al. 2016; Tilzey et al. 2006) and the use of a kite and hood (MPI 2012).

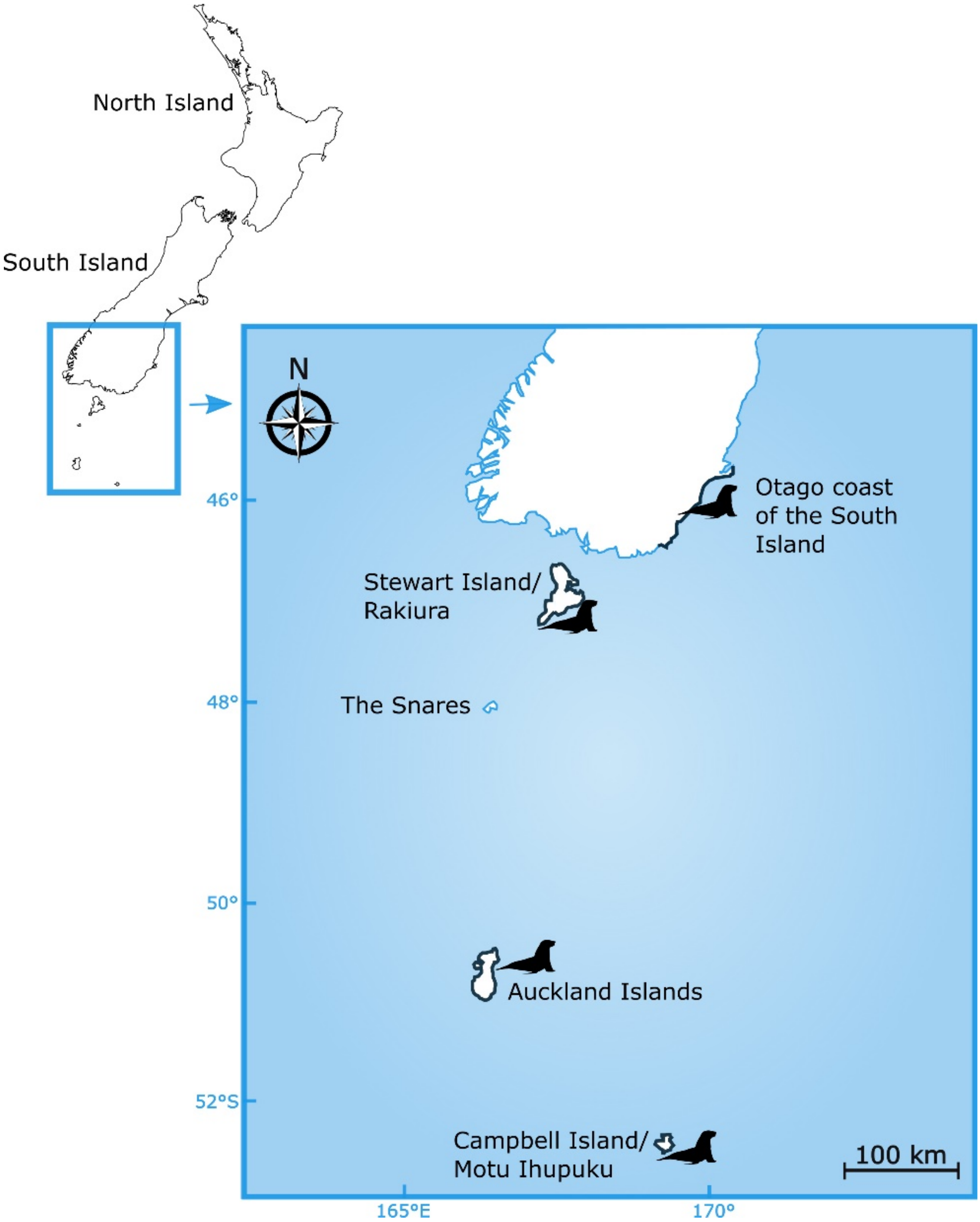

Figure 3.1: The four current NZ sea lion breeding populations located at Campbell Island, the Auckland Islands, Stewart Island and the Otago coast of the South Island, NZ. (Figure published in Hamilton & Baker (2019)).

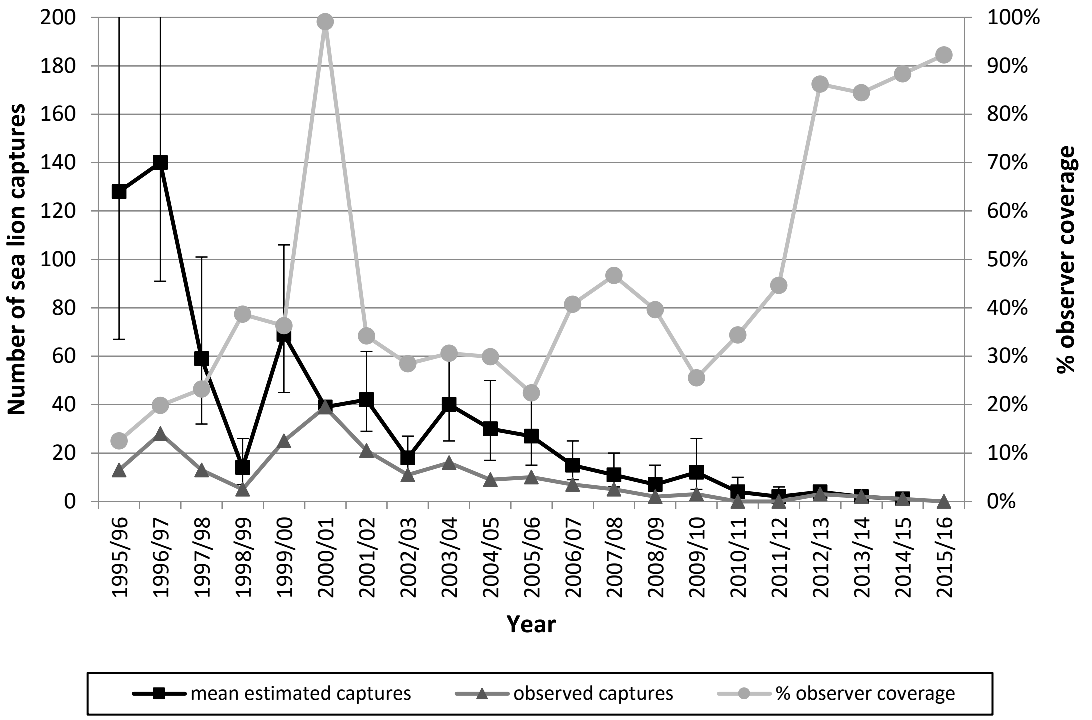

Figure 3.2: The observed number of captures (dark grey triangles) and mean estimated captures (black squares; error bars = 95% confidence interval) per year of NZ sea lions in squid trawl fisheries around the Auckland Islands from 1995/96 to 2012/13. The % observer coverage (light grey circles) is also shown. Data from 1995/96 to 2012/13 from Abraham et al. (2016); data from 2013/14 to 2015/16 from MPI (2018b).

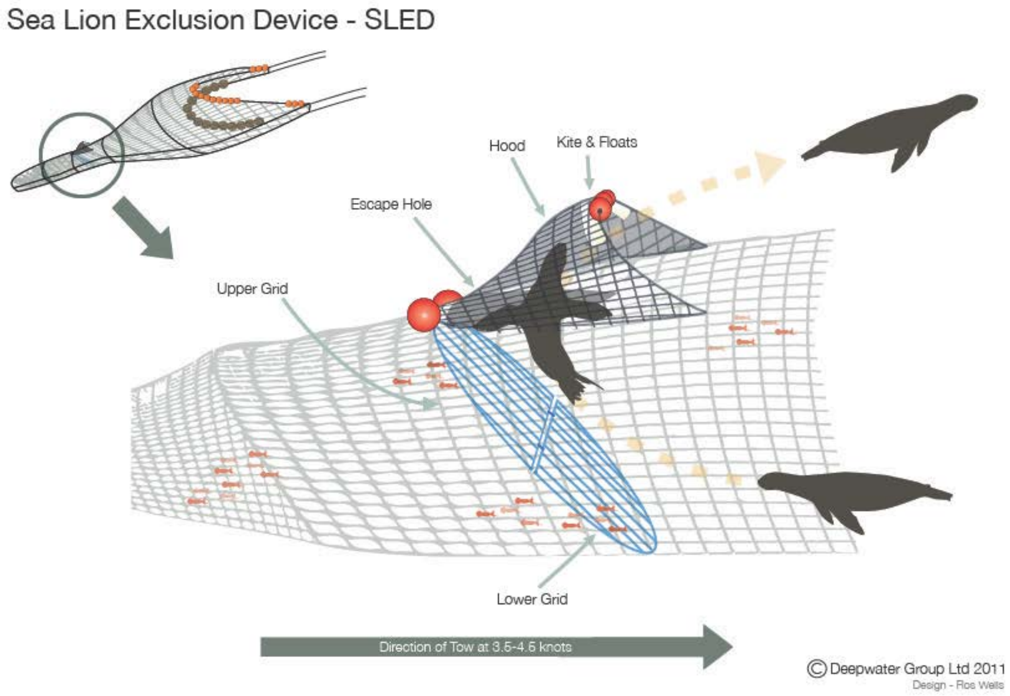

Figure 3.3: Standard Sea Lion Exclusion Device used within the Auckland Islands squid trawl fishery (figure provided by Deepwater Group Ltd, New Zealand).

Any sea lion that is caught in a trawl net that does not have a SLED fitted will die whereas SLEDs provide an opportunity for sea lions to escape trawl nets. SLEDs are thought to direct the majority of sea lions that enter trawl nets out of the net. However, a small number of sea lions are still incidentally caught, retained and hauled aboard by vessels fitted with SLEDs (Thompson et al. 2013b). These animals die within the net by drowning or trauma. Drowning may occur if the sea lion is unable to negotiate the SLED within its breath-holding ability or there is a failure with the SLED escape route such as when the hood collapses and subsequently closes the escape hole (Roe and Meynier 2012). Issues with the hood potentially closing the escape hole have been largely addressed through use of a kite and floats and by regular audit and compliance checks of the fishing fleet (e.g. Cleal et al. (2007)). Trauma could occur either from impact with the SLED grid or some other part of the trawl net system (Roe and Meynier 2012).

The implementation of the SLED as a mitigation measure has attracted controversy. Sea lion bycatch dropped substantially in the Auckland Islands squid fishery following the introduction of SLEDs, which were in widespread use by 2004/05 (Figure 3.2). However, the continued decline in the estimated sea lion population from 2004/05 to 2008/09 (when the lowest pup production was reported) led to concerns that fisheries-related mortality continued to be an issue for this species (Chilvers 2012b; Robertson and Chilvers 2011). This was based on claims that some animals that entered a net may have suffered head trauma when they collided with the SLED's hard grid that caused either immediate or post-escape mortality (Robertson and Chilvers 2011). Concerns were also raised that some sea lions may die within the net and fall out of the escape hole during hauling (Roe 2010; Roe and Meynier 2012) or that some sea lions may die after escaping through a SLED (Robertson and Chilvers 2011), both of which could lead to underestimates of bycatch.

In view of the uncertainty regarding the use of SLEDs in the Auckland Islands squid fishery, I undertook a review of the largely unpublished research and data assessments to evaluate the efficacy of SLEDs in reducing sea lion bycatch. There is benefit in ensuring that approaches developed to mitigate bycatch are effective and, if not, to move to adopt other measures to address a significant issue for sea lion conservation and for fishery management.

3.3 Methods

3.3.1 Review of research undertaken by New Zealand government and fishing industry

I reviewed all available published and unpublished literature on research that aimed to assess the efficacy of SLEDs in reducing sea lion bycatch. In particular, two areas of research were identified that, in general, addressed the following questions:

(i) Do SLEDs allow sea lions to escape from trawl nets and do these animals survive?

(ii) Is head trauma likely when a sea lion comes into contact with a stainless-steel SLED grid?

The bodies of work I reviewed included:

- an experimental approach where sea lions were deliberately trapped after passing through a SLED;
- assessments of the survivability of sea lions passing through a SLED based on reported reviews of necropsy results and video monitoring;
- attempts to obtain additional video footage of SLEDs during fishing operations;

- re-analysis of video footage from an Australian study of fur seals passing through a Seal Exclusion Device (SED) in Australia's Small Pelagic Fishery to obtain comparative data on the nature of collisions;
- a biomechanical study that simulated the impact of sea lions hitting the metal grid of a SLED; and,
- modelling of the risk of sea lions suffering mild traumatic brain injury (concussion) after striking a SLED grid.

3.3.2 Analysis of underwater video from Australian blue grenadier trawl fishery

Another component of assessing SLED efficacy is determining whether dead sea lions could drop out of the top-opening escape hole of SLEDs, which may then contribute to underestimates of bycatch levels. SEDs, with a similar design to SLEDs, have been used in the Australian blue grenadier (*Macruronus novaezelandiae*) fishery (part of the Southern and Eastern Scalefish and Shark Fishery, SESSF, Commonwealth Trawl Sector, CTS) to reduce the bycatch of fur seals, predominantly Australian fur seals (*Arctocephalus pusillus*). However, the large target fish (blue grenadier) clog the grid of standard SEDs, especially when large quantities are caught, thereby reducing the quality and quantity of fish catches as well as the efficacy of SEDs in releasing seals from nets. To address the grid clogging issue, the Australian Fisheries Management Authority (AFMA) designed an "Acoustic Release-SED" with a hinged top half 'gate' which could be held open during fishing and closed during the haul when it was thought most seals were at risk of being caught (see Figure 3A.1 in 3.7 Appendix). AFMA trialled both standard SED and Acoustic Release-SED designs on a midwater trawl vessel operating in the blue grenadier fishery off Tasmania to assist industry in improving mitigation devices and operations and to look at 'proof-of-concept' of the different SED designs. As a proxy to NZ sea lion interactions with SLEDs, 65 hours of video footage taken in 2011–2012 of Australian fur seal interactions in the Australian blue grenadier fishery was re-assessed to determine whether there was evidence that pinnipeds passively drop out of the top-opening escape hole (Hamilton et al. 2019). I incorporate the main findings from this study as part of the assessment of SLED efficacy.

3.4 Results and Discussion

3.4.1 Evidence that SLEDs allow sea lions to escape from trawl nets and that they survive

An experiment was carried out where sea lions that successfully exited the SLED were retained within an external cover net (with a small number documented on video) and later necropsied to assess whether SLEDs enabled the escape of sea lions from trawl nets and whether they survived the process (Wilkinson et al. 2003). Due to the extended periods (i.e. up to several weeks) that trawlers spend at sea each trip, it has been necessary to freeze the carcasses of caught sea lions for any later research examinations (Roe and Meynier 2012). During 1999 and 2000, six sea lions were incidentally caught in nets fitted with SLEDs and, of these, five were directed out of the SLED and retained in the cover net (Wilkinson et al. 2003). In 2001, 33 sea lions were caught in trawl nets with cover nets and, of these, 30 (i.e. 91%) passed through the SLED and were retained in the cover net, although video footage was obtained for only three animals. Although the sample size was very small, the footage indicated the three animals were likely to have survived if the cover net had not been present (Wilkinson et al. 2003). However, a veterinary pathologist who examined the retained and frozen carcasses concluded that at least one and possibly two of these animals exhibited severe internal trauma which was considered, at the time of the assessment, would have led to their subsequent death (Gibbs et al. (2001) cited in Wilkinson et al. (2003)). Necropsy assessments of all 30 animals retained and frozen in 2001 concluded that at least 55% of them had suffered trauma that would have compromised their post-exit survival (Gibbs et al. (2001) cited in Wilkinson et al. (2003)). However, Gibbs *et al.* (2003) acknowledged that freezing the carcasses may have induced changes that could be confused with true lesions. In 2008 and 2009, an experiment using five chilled and five frozen NZ fur seals (*Arctocephalus forsteri*) recovered from trawl nets (without SLEDs) in the Cook Strait hoki (*Macruronus novaezelandiae*) fishery reported that, although based on a small sample size, some lesions originally interpreted and reported to be due to trauma were indeed artefacts of freezing (Roe et al. 2012; Roe and Meynier 2012).

A re-evaluation of the 2000/01 trial when sea lions were retained in a cover net after successfully exiting a SLED (Wilkinson et al. 2003), concluded that assessing the likelihood of longer-term post-SLED survival was difficult (Roe and Meynier 2012). This was due to the inconsistencies between observations of the three video-taped sea lions, the small sample size, the uncertainties with necropsy findings as well as the inability to measure longer term survival of these animals (Roe and Meynier 2012). Middleton and Breen (2011) supported the original conclusion by Wilkinson et al. (2003) that the three videoed animals who escaped via a SLED (and retained in a cover net) were likely to have survived their interaction with the SLED as their post-SLED survival was questioned solely on the basis of necropsy results that were subsequently deemed inconclusive.

Although it would be desirable to obtain more visual evidence of NZ sea lions passing through SLEDs, there is an absence of useful video footage of SLED interactions for the Auckland Islands squid trawl fishery due to poor visibility at fishing depths as well as very low sea lion interaction rates when video footage was obtained. Approximately 600 hours of video footage has been collected primarily to assess SLED deployment and engineering characteristics (R. Wells, Deepwater Group NZ, personal communication). I reviewed a sample of this video footage and found it was largely impossible to observe SLED efficacy at fishing depths as visibility was very poor due to lighting limitations, water depth, fine debris and squid ink suspended in the water column, as well as sea lion interaction rates being very low.

Necropsy data have been collected from sea lions incidentally caught in trawl fisheries since 1996/97. Roe and Meynier (2012) undertook a review of necropsy data for sea lions recovered from nets with (n = 98) and without (n = 50) SLEDs as well as a further 15 carcasses that had no information on whether a SLED was used or not. Although the aim of necropsies undertaken from 1996/97 to 1999/2000 was to obtain morphometric information and not to assess the types or severity of injuries (Roe 2010), a consistent method of trauma classification criteria was applied to all past data (Roe and Meynier 2012). Roe and Meynier (2012) acknowledged that any assessment of long-term necropsy data was compromised by the understanding that some lesions originally thought to be due to trauma were artefacts of freezing. However, necropsy assessments for all 163 animals concluded that all had died as a result of drowning (Roe 2010; Roe and Meynier 2012). Furthermore, the data indicated that the overall reported trauma severity, the prevalence of apparent head bruising and/or patterns of bruising involving the sternum, shoulders and axillae appeared to be unrelated to whether or not a SLED was fitted to the trawl net (Roe and Meynier 2012). In addition, many injuries that were observed on carcasses were thought to have occurred well before death (Gibbs et al. 2003; Roe and Meynier 2012). It is also known that, prior to freezing, dead animals have been handled poorly on board vessels with reports that carcasses were often moved considerable distances through the vessel and, by necessity, dropped up to 8 m into vessel holds as they could not pass through product handling systems due to their size (R. Wells, Deepwater Group NZ, personal communication).

Middleton and Breen (2011) undertook a review of veterinarian evaluations of necropsy assessments from incidentally caught sea lions and reiterated the findings of Roe (2010) and Roe and Meynier (2012) who established that a number of lesions previously classified as ‘acute blunt trauma’ were most likely artefacts from freezing the carcasses. Supported by reviews of original necropsy data by Roe and Meynier (2012), the cause of death for all necropsied sea lions remained as drowning and not the direct effects of trauma injuries. However, Middleton and Breen (2011) identified that there was an increasing focus on the possibility of head trauma during several veterinary assessments that aimed to inform the likelihood of sea lions surviving overall trauma if the animal had not drowned in the net (e.g. Roe 2010; Roe and Meynier 2012). Despite indications that head lesions could equally also be artefacts of carcass freezing, necropsy assessments scored such lesions conservatively and considered that any indication of head injury had the potential to reduce survival through reduced consciousness (Middleton and Breen 2011).

3.4.2 Evidence of a low risk of significant head trauma when sea lions come into contact with stainless steel SLED grids

In the absence of useful video footage of NZ sea lions interacting with SLEDs, the NZ government and other squid fishery stakeholders considered that the behaviour and responses of fur seals to SED interactions may provide information to help assess the possible nature of sea lion and SLED interactions and, in particular, the potential of head trauma injuries that may result from head-first collisions with a metal grid (MPI 2012). Therefore, a review was undertaken (Lyle 2011) of video footage from a 2006/07 study of fur seals interacting with SEDs deployed in the Australian Small Pelagic Fishery (Lyle et al. 2016). The nature (i.e. whether seals struck the grid, the speed at which they struck and where on the grid the impact occurred) and potential consequences of collisions with a rigid steel grid were reviewed (Lyle 2011). Interactions with the SED were described for 132 seals, although the clarity and quality of the video footage influenced how much information could be obtained for each interaction (Lyle 2011). About one third of the seals that entered via the mouth of the trawl approached the SED head-first and most of them experienced a head-first collision with the grid (usually the upper half of the SED grid) with the angle of the head usually more or less perpendicular to the grid (Lyle 2011). Impact speeds were estimated with head-first impacts from the first interaction a seal had with the SED occurring at a slightly faster speed (average 3.5 m s^{-1}, range 2.9–6.1 m s^{-1}) than subsequent head-first collisions (Lyle 2011). Impact speeds were a function of trawl speed and seal swimming speed, with trawl speeds ranging 1.5–2.6 m s^{-1} (3–5 knots), suggesting that some seals were gliding rather than actively swimming within the net (Lyle 2011). There was no significant difference in the mortality rates between seals that had at least one head-first collision with the SED grid and those that did not contact the SED head-first (Lyle 2011). It was considered that the likely speed and location of collisions that were inferred and the estimated collision speeds were consistent with the observed swimming speeds for NZ sea lions (MPI 2012). Lyle (2011) did not discuss the implications of the video footage assessment on the extent and nature of impact injuries or the subsequent survival of fur seals as there had been no post-interaction or post-mortem examination of the seals during the original 2006/07 study.

Biomechanical modelling to estimate the impact of head-first collisions between sea lions and SLED grids (MPI 2012; Ponte et al. 2010) indicated that it was extremely unlikely that an impact with a SLED would cause brain trauma at a level to cause death (Abraham 2011). To assess the likelihood of a sea lion acquiring a life-threatening brain injury or mild traumatic brain injury (i.e. concussion) as a result of a head impact with a SLED stainless steel grid, Ponte *et al.* (2010) used a validated method for measuring head impact injury in human pedestrians ('crash tests') with scaling and extrapolating to account for the relative head and brain mass of a sea lion. For particular impact locations on the grid, the likelihood of a life-threatening brain injury, based on swim and collision speed and effective sea lion head mass, was determined using the 'crash test' results with results indicating that a sea lion impacting with the grid may incur some sort of brain injury (Ponte et al. 2010). Ponte *et al.* (2010) summarised a scenario where a female sea lion had a 10 m s^{-1} collision with the SLED grid at the stiffest location, which indicated the likelihood of a life-threatening brain injury may be higher than 85%. However, this impact speed is likely to represent the worst case scenario, especially if Lyle's (2011) fur seal interaction speeds (average 3.5 m s^{-1}, range 2.9 - 6.1 m s^{-1}) are considered indicative of NZ sea lion interactions, and may be more dependent on individual sea lion behaviour than the grid design (Industrial Research Ltd. (Banerjee 2011). It was considered that a more realistic scenario would be to estimate the probability of a fatal impact and then design the SLED accordingly (Industrial Research Ltd. (Banerjee 2011). Using results from the 'crash-test' research (Ponte et al. 2010) and the re-analysis of video footage of fur seals interacting with SEDs (Lyle 2011), Abraham (2011) modelled the probability that a sea lion interacting with a SLED suffers concussion. The modelling simulations included parameters such as the location that animals collide with the SLED grid and the speed of impact, as well as data on the typical number of head collisions between fur seals and the grid (Abraham 2011). This study sought to test the likelihood of the conditions (speed, orientation, number of impacts, area of the grid) needed to induce concussion and concluded that the probability of a concussion that could result in the animal drowning after exiting the SLED was very unlikely to exceed 10% (Abraham, 2011). There are a number of limitations in this modelling work when considering sea lion interactions with the Auckland Islands squid fishery. These include the unknown level of scaling of the simulated risk of brain injury predictors to actual values experienced by sea lions, the use of data derived from humans to calculate concussion probability, and the use of data from fur seals in an Australian fishery to obtain grid collision information (Abraham 2011).

3.4.3 Lack of evidence that dead pinnipeds passively drop out from top-opening escape holes

The assessment of underwater video footage from Australia's blue grenadier fishery was limited by a lack of metadata to enable interpretation of footage in context of operational factors coupled with a range of variables during trials (e.g. day versus night trawls, weather conditions and trials deploying two different SED designs) confounding the assessment (Hamilton et al. 2019). In addition, Australian fur seal interaction rates were low during the 2011 and 2012 trials further limiting data robustness. However, data from 31 trawl shots in July 2011 and June 2012 were supplemented by video observations of seal interactions recorded by AFMA during 2009 trials.

Of 49 seals observed, 26 were only seen outside the net, five partially entered the net via the escape hole with most taking blue grenadier that were stuck on the SED grid, and 18 seals were captured in the net. Of the 18 captured seals, one entered and exited via the SED escape hole, one exited via the net mouth, six were thought to enter via the net mouth and exit the SED escape hole, one was brought on deck and released alive, one had an 'unknown fate' and eight died (see Table 3A.1 in 3.7 Appendix). All eight dead seals were recorded as bycatch once the net was hauled on deck. However, five of these were from trials of the modified Acoustic Release-SED design which had half of the SED grid held open during setting and at fishing depth (see 3.7 Appendix) and, as all five entered the net during setting, they swam through the open grid and their carcasses were retained in the codend. While there was evidence that live seals were able to negotiate and exit from a top-opening escape hole fitted with a hood, there was no evidence of seals having a hard impact with SED grids that would compromise their survival, or that dead seals fell out of top-opening SED escape hole (Hamilton et al. 2019).

3.5 Summary and future directions

Since the implementation of mitigation techniques, including the widespread deployment of SLEDs, there has been an encouraging reduction in the sea lion bycatch reported for the Auckland Islands squid trawl fishery (Figure 3.2; Thompson *et al.* 2013). With continued commitment to mitigation, and maintaining adequate observer coverage, further reductions in estimated bycatch levels should be achievable.

It is considered that the risk of sea lions dying in the net and falling out of the top-opening escape hole and, therefore, not being included in reported fisheries-related bycatch observations is unlikely due to the SLED configuration, primarily the fitting of a hood on a top-mounted escape hole (MPI 2012). Dead pinnipeds are also observed to be negatively buoyant and, therefore, would not float out a top-opening escape (M. Gerner, AFMA, unpublished data). This is further supported by the observations from the Australian blue grenadier fishery which uses a top-opening SED (Hamilton et al. 2019). However, further work is required to better quantify potential cryptic mortality by undertaking targeted research using underwater video assessments of trawl nets fitted with SEDs or SLEDs.

While there is evidence to suggest SLEDs reduce the numbers of animals trapped by increasing the number that can escape, there have been suggestions that SLEDs may be ineffective in reducing sea lion mortality because some animals may die later from injuries, particularly head trauma. However, the extensive efforts to test the efficacy of SLEDs have shown that most sea lions are likely to survive following their escape from a trawl net via a SLED. Future research to address any remaining uncertainty regarding the post-SLED survival of sea lions could focus on the use of technology to remotely record animals that pass through a SLED, followed up with assessments of the survival of these animals. Technological options could include the use of acoustic tags fitted to sea lions and receivers installed on the escape holes of SLEDs as suggested by Bradshaw et al. (2013). However, such work would require the handling of large numbers of animals, substantial financial investment and, depending on the technology used, may require assessments on the impact of this research on target species catch rates.

It is likely that there are other factors that are now primarily responsible for the continued reported decline of the NZ sea lion population at the Auckland Islands. Disease events have been reported at sea lion breeding colonies over the last 20 years that have affected adult females and their pups. Epizootic events affecting pup production were reported for the 1997/98, 2001/02 and 2002/03 seasons, with adult female mortality also occurring during the 1997/98 event (Castinel et al. 2007; Chilvers 2012b; DoC and MPI 2016; Wilkinson et al. 2003; Wilkinson et al. 2006). Population modelling, taking into account infrequent epizootic events (once every 15 years) and current levels of fisheries bycatch, indicated that, even under these pressures, there would be slow population growth of the sea lion population at the Auckland Islands (Hamilton and Baker 2016a). However, there are recent indications that both the frequency and magnitude of epizootic events may have been underestimated due to researchers leaving the Auckland Islands before pups go to sea (S. Childerhouse, personal communication). Recent modelling has indicated that the continued NZ sea lion population decline at the Auckland Islands is unlikely to be related to direct fisheries impacts (with reported bycatch of adult animals), but more likely linked to decreased breeding productivity and mortality in the first year or two of life (Hamilton and Baker 2016a; Roberts and Doonan 2016). It is also thought that food limitation and shark predation effects may be particularly crucial for the Auckland Islands population, but these factors are poorly understood (Roberts and Doonan 2016).

The research and assessments reviewed here indicate that SLEDs have contributed to the reduction in NZ sea lion bycatch in the Auckland Islands squid fishery and that most sea lions are likely to survive subsequent to their escape from a trawl net via a SLED. Furthermore, recent modelling has concluded that current levels of fisheries-related mortality is unlikely to be the main driver of the continued decline in the sea lion population at the Auckland Islands (see Chapter 4 of this thesis, (Hamilton and Baker 2016a; Roberts and Doonan 2016). Therefore, future work may be better directed towards alternative research and management areas that may be more effective in addressing issues to reverse NZ sea lion population decline. This may include research into the frequency, magnitude and mitigation of epizootic events, the scale and importance of changes in food availability, and management actions that increase pup survival on the colony and in the first couple of years at sea.

3.6 Appendix: Review and analysis of underwater video of seal interactions in blue grenadier trawl nets fitted with SEDs (full report in Hamilton et al. (2019))

A standard top-opening Seal Exclusion Device (SED; Figure 3A.1) is deployed in the Australian blue grenadier trawl fishery (Australian Fisheries Management Authority (AFMA) 'SESSF Gear Directions'). However, the large target fish (blue grenadier) clog the grid of standard SEDs, thereby reducing the quality and quantity of fish catches as well as the efficacy of SEDs in allowing seals to exit from nets. To address the grid clogging issue, AFMA designed an “Acoustic Release-SED” with a hinged top half ‘gate’ and a fixed lower half grid (Figure 3A.2). Blue grenadier are predominantly caught at 300–600 m (Tilzey 1994 cited in Hamer and Goldsworthy (2006)) whereas Australian fur seals dive < 200 m (190 m for seals entering the net during shooting, Hamer and Goldsworthy (2006)). During setting and when fishing at depths beyond the diving range of seals, the top half of the Acoustic Release-SED grid is held open by an acoustic actuator. This allows the large target fish to flow smoothly into the codend and, additionally, the open grid ‘gate’ covers the SED’s top-opening escape hole reducing fish loss. Before hauling through seal diving range depths, the ‘gate’ is triggered to close by an on-board acoustic transponder deployed by hand over the stern of the vessel. This sends a signal to the release device (sewn into the net) which frees the latch allowing the gate to drop. Therefore, while hauling through seal diving range depths, seals are excluded from the codend and any that enter the net can escape via the SED top-opening hole (Hamilton et al. 2019).

From 2008–2012, AFMA trialled the standard SED and the Acoustic Release-SED on a midwater factory trawl vessel fishing in the winter blue grenadier fishery off Tasmania. Underwater video footage was obtained during these trials to look at proof-of-concept of the different SED designs and assist industry in improving mitigation devices and operations. A project was undertaken to reassess available video footage to characterise the behaviour and fate of fur seals interacting with trawl gear, and to assess the function of the different SED designs (Hamilton et al. 2019). A summary of the seal interactions from 2009, 2011 and 2012 video footage is provided in Table 3A.1.

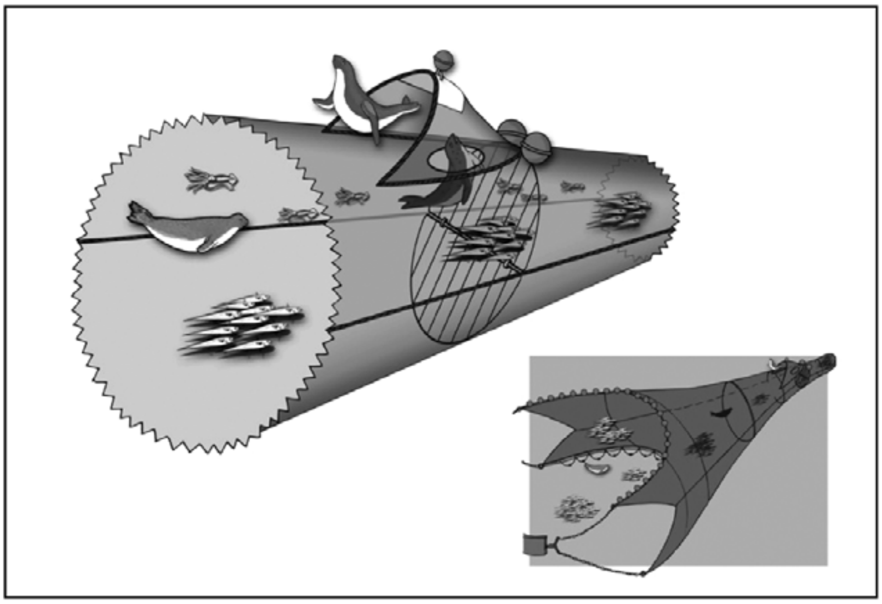

Figure 3A.1: The standard Seal Exclusion Device (SED) used in blue grenadier factory midwater trawlers off western Tasmania. Image from Hoki Fishery Management Company, NZ published in Hamer and Goldsworthy (2006).

A

B

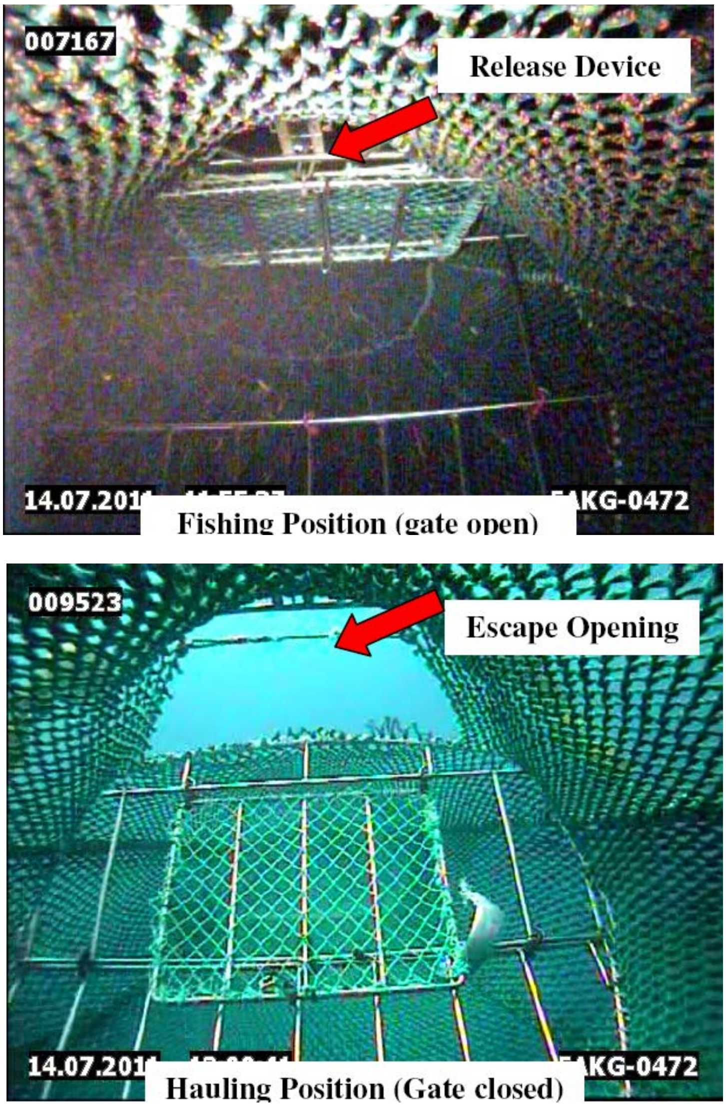

Figure 3A.2: Images of the Acoustic Release-SED with the top 'gate' held open (A) and, following release using the on-board acoustic transponder, closed for the haul (B). Images supplied by AFMA.

Table 3A.1: Summary of seal interactions observed using underwater video footage taken in 2009, 2011 and 2012 from trials of Seal Exclusion Devices (SED) in the Australian blue grenadier trawl fishery.

<table>
<tr><th colspan="2">Description</th><th>Number</th></tr>
<tr><td colspan="2">Number of shots with video footage</td><td>49</td></tr>
<tr><td colspan="2">Total number of shots with seal interaction(s)</td><td>30</td></tr>
<tr><td colspan="2">% of shots with seal interaction</td><td>61%</td></tr>
<tr><td colspan="2">Minimum number of seals* seen (all shots)</td><td>49</td></tr>
<tr><td rowspan="2">Seal only observed outside net (n > 26)</td><td>Seal outside net during shot</td><td>>12</td></tr>
<tr><td>Seal outside net during haul</td><td>14</td></tr>
<tr><td>Seal partially enters net and survives (n = 5)</td><td>Partially enters via SED escape hole, and exits usually with a fish</td><td>5</td></tr>
<tr><td rowspan="5">Seal observed inside net and survives (n = 9)</td><td>Enters SED escape hole during haul and exits escape hole</td><td>1</td></tr>
<tr><td>Enters via net mouth and assume exits via net mouth</td><td>1</td></tr>
<tr><td>Enters via net mouth and observed exiting SED escape hole</td><td>4</td></tr>
<tr><td>Enters via net mouth and assume to exit SED escape hole</td><td>2</td></tr>
<tr><td>Enters net near end of haul, retained in net and released alive on deck</td><td>1</td></tr>
<tr><td rowspan="3">Seal observed in net and dies (n = 8)</td><td>Dead in hood**</td><td>1</td></tr>
<tr><td>Enters via net mouth, dies at SED grid</td><td>2</td></tr>
<tr><td>Enters via net mouth during shot, passes through open SED grid and dies in codend</td><td>5</td></tr>
<tr><td>Seal observed in net, unknown fate (n=1)</td><td>In net, unknown fate</td><td>1</td></tr>
</table>

*some seals seen more than once over a period of minutes and have assumed it is the same individual.
**one seal dead and recorded as being retained in hood when net hauled onboard although no footage of this trawl shot was obtained as battery flat on camera.

Chapter 4 — Current fisheries bycatch levels unlikely to be the main driver of New Zealand sea lion (*Phocarctos hookeri*) population trajectory at the Auckland Islands

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Updated and developed in the context of my thesis framework, with models expanded to include emigration, inbreeding depression, and recent bycatch data – no change to overall findings or conclusions.

4.1 Abstract

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

Population Viability Analysis (PVA) modelling, using VORTEX, of the Auckland Islands NZ sea lion population was undertaken to ascertain if the reported levels of bycatch of sea lions in trawl fisheries around the Auckland Islands are sustainable following substantial and effective mitigation to reduce bycatch, particularly in the Auckland Islands squid fishery. Modelling indicated slow population growth of the Auckland Islands NZ sea lion population with current bycatch estimates from all Auckland Islands trawl fisheries. Additional modelling seeking explanations for observed population

declines since the late-1990s indicated that epizootic events that reduce pup production may have a greater impact on population growth, especially if these events are more frequent than previously assumed. These results suggest that sea lion bycatch in the squid fishery and other trawl fisheries around the Auckland Islands is unlikely to be currently having a significant impact on the Auckland Islands' NZ sea lion population. Therefore, resources should be directed towards other hypotheses for any future negative impact on the NZ sea lion population trajectory as well as continued refinement of mitigation techniques to reduce fisheries-related mortality.

4.2 Introduction

The New Zealand (NZ) sea lion, *Phocarctos hookeri*, a NZ endemic, has a Nationally Vulnerable status on the NZ Threat Classification system (Baker et al. 2019). It was previously classified as 'Vulnerable' on the IUCN Red List in 2008 based on a 30% decline in pup production at some of the major breeding colonies from 1998–2008 (Gales 2008) (Figure 4.1). It was uplisted to 'Endangered' on the IUCN Red List in 2015 (Chilvers 2015) based on modelling published in 2012 which predicted ongoing population decline following a major decrease in numbers from the late-1990s to late-2000s at the main population on the Auckland Islands (Chilvers 2012b). The species once occurred from the northernmost cape of NZ's North Island to sub-Antarctic Campbell Island (Childerhouse and Gales 1998) (see Figure 3.1). However, historically, subsistence hunting followed by commercial sealing greatly reduced both the population and the breeding distribution (Childerhouse et al. 2010b).

The species' birthing season occurs from mid-December to early January each austral summer[2] with breeding male and female sea lions likely to be ashore for prolonged periods between late November and January (MPI 2017a). Pregnant females give birth to a single pup in late December, stay ashore for about 10 days after giving birth and then alternate between foraging trips and returning to suckle their pups for a further 8–12 months before weaning (MPI 2017a). Based on 2014/15 pup production estimates from the Auckland Islands (Childerhouse et al. 2017) and Campbell Island (Childerhouse et al. 2015), 68% of all sea lion pups are born at the Auckland Islands with 30% at Campbell Island. The remainder are born at Stewart Island and the Otago coast of NZ's South Island and, occasionally, The Snares and Codfish Island (DoC and MPI (2016); see Figure 3.1).

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

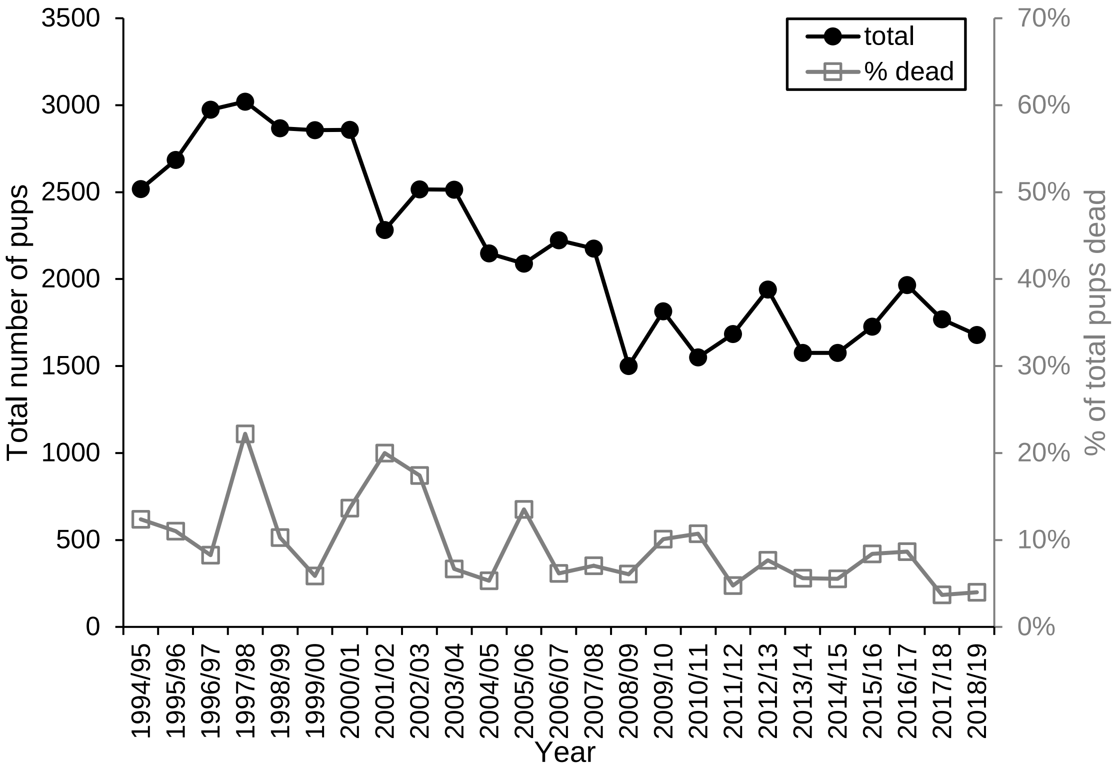


Figure 4.1: Total estimated pup production and the proportion of those recorded dead during annual mark/recapture estimate field work (late December to mid-February) for NZ sea lions at the Auckland Islands 1994/95–2016/17. Data sources: pre 2012/13 Chilvers (2012c); 2012/13–2016/17 Childerhouse et al. (2017); 2017/18 Childerhouse et al. (2018); 2018/19 Dodge et al. (2019).

Commercial trawl fisheries have been implicated in the observed decline of NZ sea lions owing to the incidental mortality (hereafter referred to as “bycatch”) of sea lions in trawl nets (Robertson and Chilvers 2011). Over the past couple of decades, most fisheries-related mortality of sea lions has occurred in the Auckland Islands squid fishery (Thompson et al. (2013b); Figure 4.2; Table 4.1). However, mortality has also been recorded or estimated in the Auckland Islands scampi fishery, Auckland Islands non-squid/scampi trawl fisheries, the southern blue whiting fishery operating near Campbell Island and the Stewart-Snares shelf trawl fisheries (Abraham et al. 2016; Thompson et al. 2013b); Table 4.1). The sea lions caught in the southern blue whiting trawl fishery are considered to emanate from the Campbell Island population owing to the proximity of this fishery to Campbell Island and because no tagged animals from Auckland Island colonies have been observed killed in the fishery to date (Department of Conservation, unpublished data). Recent satellite tracking research has also shown that sea lions from the Campbell Island population typically forage in close proximity to Campbell Island (M.-A. Lea, unpublished data). However, occasionally Auckland Island animals may be impacted in this fishery as there is limited evidence that some male sea lions from the Auckland Islands forage around Campbell Island and may overlap spatially with the southern blue whiting trawl fishery near Campbell Island (Geschke and Chilvers 2009; Maloney et al. 2012). Satellite tracking data also showed one nursing female sea lion from the Auckland Islands undertaking one return trip to Campbell Island (Simon Childerhouse, personal communication). Only one sea lion (a female in March 2005) caught in the Stewart-Snares shelf trawl fisheries has had an identifying brand or tag (Thompson et al. 2015) and it originated from Sandy Bay, Enderby Island (Ministry for Primary Industries (MPI, NZ), unpublished data); although the provenance of other animals caught in this fishery is unknown.

The annual Auckland Islands squid fishery (targeting *Nototodarus sloanii*), one of NZ's largest, more valuable fisheries, uses a combination of bottom and mid-water trawls operating at bottom depths of about 150–250 m. Although not completely understood, the foraging areas and depths of NZ sea lions have been shown to overlap, in some areas, with commercial trawl fishing activity in the Auckland Islands squid fishery, with the foraging areas of females shown to have more overlap with fishery operations than those of males (Chilvers 2008a; Chilvers 2009; Chilvers et al. 2011; Leung et al. 2013; Leung et al. 2012). Peak activity in the fishery occurs between February and May (MPI 2017a) coinciding with part of the lactation period for breeding female sea lions. Sea lion mortality in the Auckland Islands squid fishery has been monitored by government observers since 1988 (Wilkinson et al. 2003), although observer coverage has varied from <10% to 99% over this period (Figure 4.2). The mean estimated level of sea lion bycatch peaked in the mid to late 1990s, specifically at 128 in 1995/96 and at 140 in 1996/97 (Abraham et al. (2016); Figure 4.2). To reduce the impact of fisheries-related mortality on the sea lion population, management measures have included the establishment of a 12 nautical mile marine reserve around the Auckland Islands excluding all fishing within that range, the instigation of mortality limits that can trigger spatio-temporal closures, and the development and implementation of a 'Sea Lion Exclusion Device' or SLED (Hamilton and Baker 2015a; MPI 2012). The SLED comprises an additional section of netting inserted between the lengthener and codend of the trawl net with an angled two or three part metal grid that aims to direct sea lions to an escape hole in the top of the net and exclude them from the trawl codend (Hamilton and Baker 2015a; MPI 2017a; Wilkinson et al. 2003). Since 2004/05, all vessels in the Auckland Island squid fishery have used government-specified, standardised SLEDs (Hamilton and Baker 2015a; MPI 2012; MPI 2017a). Following widespread SLED use, the annual mean capture estimates of sea lions in the Auckland Islands squid fishery declined from 30 (17–50 95% CI, 30% observer coverage) in 2004/05 and have been < 5 since 2010/11 (95% CI ranging from 0–10), with high observer coverage (Abraham et al. 2016; MPI 2018b) (Figure 4.2; Table 4.1). However, the annual sea lion population estimates (based on estimates of pup numbers) continued to decline to the lowest reported estimate of 1501 pups in 2008/09 (Childerhouse et al. 2017; Chilvers 2012c; Chilvers et al. 2007; Robertson and Chilvers 2011) (Figure 4.1). This led to uncertainty regarding the efficacy of SLEDs with claims, in particular, that some animals could suffer head trauma from hitting the SLED's hard grid that may compromise their post-escape survival (Robertson and Chilvers 2011). Although the research on the effectiveness of SLEDs has been complex, concerns surrounding the efficacy of SLEDs in allowing most sea lions to exit a trawl net and survive have been shown to be largely unfounded (Hamilton and Baker 2015a). Nevertheless, the impact of the Auckland Islands squid fishery, in particular, on the sea lion population has continued to be a contentious issue particularly given additional concerns based on an apparent increased female bias in bycatch (Robertson and Chilvers 2011).

The other fishery recording noteworthy levels of interactions with sea lions around the Auckland Islands has been the Auckland Islands scampi (*Metanephrops challenger*) fishery which utilises light bottom trawl gear operating at 200–500 m on the continental slope (MPI 2013). This fishery, which does not deploy SLEDs, recorded annual mean capture estimates of 6–10 sea lions for the last five years of reported data; i.e. 2008/09 to 2012/13 (Abraham et al. 2016) (Table 4.1). The Auckland Islands non-squid/scampi trawl fishery, which primarily targets orange roughy (*Hoplostethus atlanticus*) and hoki (*Macruronus novaezelandiae*), has recorded occasional interactions with sea lions in the past although there have been no estimated captures since 2004/05 (Thompson et al. 2013b) (Table 4.1).

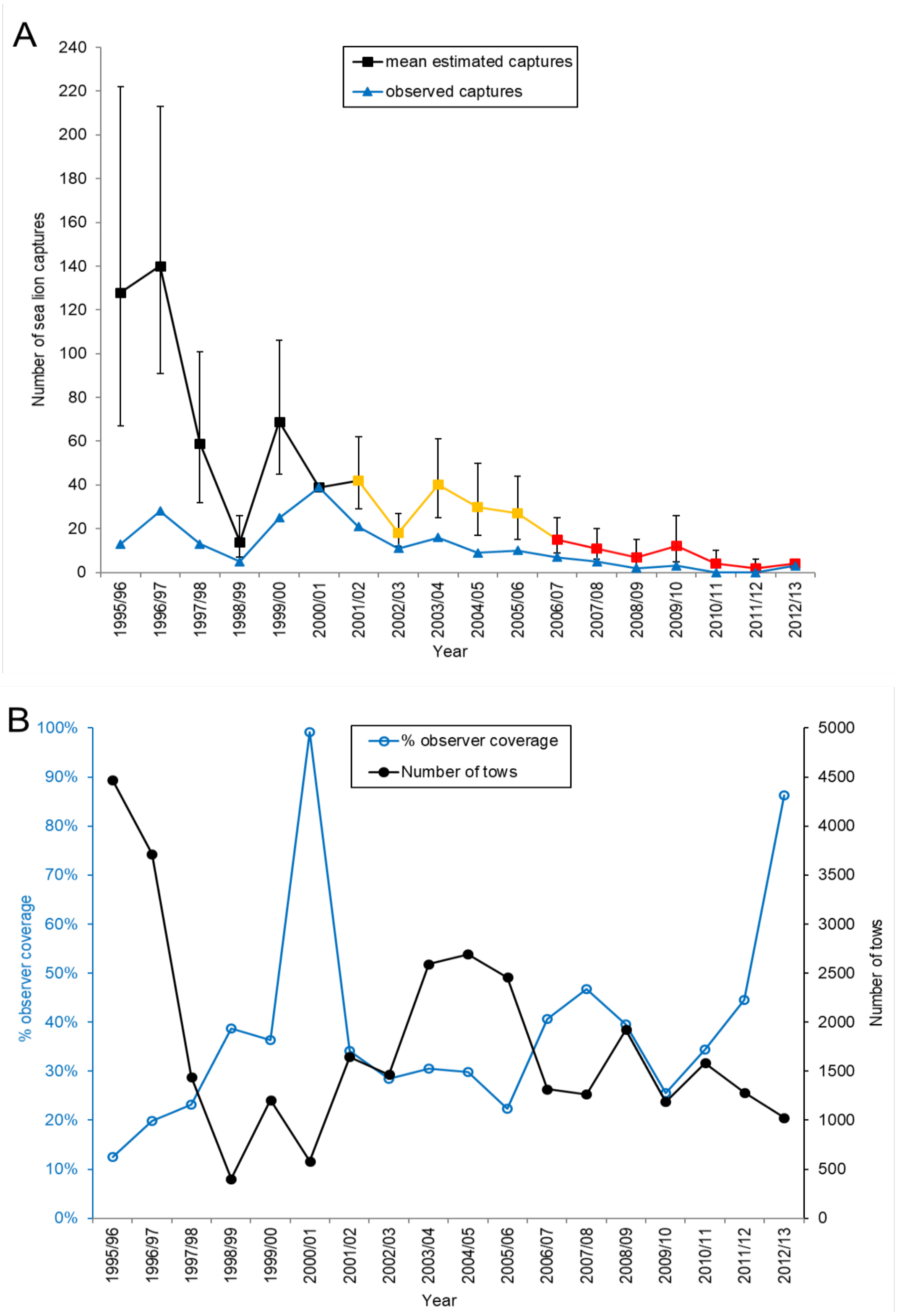


Figure 4.2: For the Auckland Islands squid trawl fishery from 1995/96 to 2012/13: A) the observed number of captures (blue triangles) and mean estimated captures with 95% confidence intervals (black squares before SLEDs, yellow squares during SLED implementation and refinement, red squares with standardised SLEDs deployed), and B) the % observer coverage (blue open circles) and number of tows (black circles). Data for 1995/96 to 2012/13 from Abraham et al. (2016).

Table 4.1: NZ sea lion capture data from New Zealand trawl fisheries from 2006 to 2013. Data from Abraham et al. (2016).

Fishery and Location	Target species	Year	Total annual trawl effort (tows)	Observer coverage (%)	Number of observed captures	Mean estimated captures (95% CI)
Southern blue whiting trawl fishery (SBW6I); Campbell Islands	southern blue whiting (*Micromesistius australis*)	2007	615	35	6	18 (8-32)
		2008	814	40	2	5 (2-11)
		2009	1205	25	0	1 (0-7)
		2010	1114	36	11	25 (16-38)
		2011	1223	36	6	15 (8-25)
		2012	893	74	0	0 (0-1)
		2013	773	100	21	21 (21-22)
Stewart-Snares shelf trawl fisheries; southern end of the Stewart-Snares Shelf area	primarily squid but also hoki (*Macruronus novaezelandiae*), jack mackerel (*Trachurus* spp.) and barracouta (*Thyrsites atun*)	2006/07	15,023	9	1	3 (1-7)
		2007/08	12,488	12	1	3 (1-6)
		2008/09	11,054	15	0	2 (0-5)
		2009/10	12,434	16	1	2 (1-5)
		2010/11	10,753	12	0	1 (0-4)
		2011/12	11,834	15	1	2 (1-4)
		2012/13	11,581	26	1	2 (1-4)
Auckland Islands scampi trawl fishery; Auckland Islands	scampi (*Metanephrops challengeri*)	2006/07	1328	8	1	9 (3-16)
		2007/08	1327	7	0	8 (2-15)
		2008/09	1457	4	1	10 (3-18)
		2009/10	- *	- *	0	5 (1-11)
		2010/11	1401	15	0	7 (2-15)
		2011/12	- *	- *	0	7 (2-15)
		2012/13	1093	12	0	6 (1-13)
Auckland Islands non-squid/scampi trawl fishery; Auckland Islands	orange roughy (*Hoplostethus atlanticus*) and hoki	2006/07	38	5	0	0 (0-1)
		2007/08	147	45	0	0 (0-2)
		2008/09	121	50	0	0 (0-2)
		2009/10	77	66	0	0 (0-1)
		2010/11	131	37	0	0 (0-2)
Auckland Islands squid trawl fishery (SQU6T); Auckland Islands	arrow squid (*Nototodarus sloanii*)	2006/07	1317	41	7	15 (9-25)
		2007/08	1265	47	5	11 (6-20)
		2008/09	1925	40	2	7 (2-15)
		2009/10	1188	26	3	12 (5-26)
		2010/11	1583	34	0	4 (0-10)
		2011/12	1281	45	0	2 (0-6)
		2012/13	1027	86	3	4 (3-6)

* some effort data not available due to anonymity requirements

There have been concurrent ‘natural’ impacts on the Auckland Islands sea lion population that have added to the complex nature of understanding the effect of incidental fisheries-based mortality. Epizootic events resulted in the deaths of 53%, 32% and 21% of pups produced for the 1997/98, 2001/02 and 2002/03 seasons (Figure 4.1), respectively, with additional adult female mortality also occurring during the 1997/98 event (Castinel et al. 2007; Chilvers 2008b; Chilvers 2012b; DoC and MPI 2016; Wilkinson et al. 2003). In 2008/09, a 31% drop in pup production in one year (to 1501 pups, Figure 4.1) was attributed to females not returning to breed in that year although the cause of this was not established (Robertson and Chilvers 2011). Researchers have typically undertaken mark-recapture studies of the sea lion population on the Auckland Islands from late December to mid-February every year. However, there is recent evidence that disease events may be occurring on colony after this time and, therefore, pup mortality from epizootics may have been underestimated (Simon Childerhouse, personal communication; Childerhouse et al. (2016)).

To inform Auckland Island squid fishery management, modelling has been undertaken over a number of years to estimate sea lion population projection and evaluate the population consequences of alternative mortality control rules (Breen et al. 2010). Chilvers (2012b) undertook a Population Viability Analysis (PVA) that predicted the ‘functional extinction’ (i.e. ‘quasi-extinction’ set at 1000 animals) of the Auckland Islands sea lion population in less than 100 years from 1995 and concluded that the level of bycatch from trawl fisheries around the Auckland Islands was the most significant known negative impact on the sea lion population. This modelling, used to inform the up-listing of NZ sea lions to ‘Endangered’ on the IUCN Red List in 2015 (Chilvers 2015), did not consider the effectiveness of mitigation, particularly over the last ~20 years, that has resulted in fisheries-related mortality being greatly reduced (Abraham et al. 2016) (Figure 4.2). Therefore, the conclusion that fisheries-based mortality continues to be the factor driving sea lion population decline (Chilvers 2012b; Chilvers 2015) may no longer be valid and other factors may be limiting population growth at the Auckland Islands.

Clarification of the possible impacts on the observed sea lion population trajectory is necessary if effective management actions are to be developed to increase population growth for this species. For this reason, further assessment of the impact of the Auckland Islands squid fishery, as well as other trawl fisheries around the Auckland Islands, on the viability of the sea lion population, is needed. Taking into account the efficacy of SLEDs and other mitigation measures, the aim of this chapter is to undertake a PVA of the Auckland Islands sea lion population to understand the key demographic factors driving trends in the population and enable evaluation of current levels of fisheries-related mortality.

4.3 Methods

A population model was developed based on the most robust population dynamics data for the NZ sea lion. To promote transparency, the program VORTEX, Version 10.2.17.0 (Lacy and Pollak 2017) was used as it is freeware that is easily accessed and understood and is widely used for undertaking PVAs in a range of situations (Hamilton and Moller 1995; Midwood et al. 2015; Prowse et al. 2013). This program simulates survival and reproductive events in successive years for each individual in a population by the Monte Carlo method. It is stochastic in that it imposes variations in annual survival and reproduction by random number generations according to prescribed probability distributions for reproduction and survival rates.

Parameter values used in the modelling

The PVA modelling was developed using published data for NZ sea lions from a NZ Department of Conservation long-term demographic study on the Auckland Islands population. It was assumed that sea lions killed in the Auckland Islands fisheries were from the Auckland Islands breeding colonies. The input demographic parameters are summarised in Table 4.2 with further information below. Each model was run for a 30-year period with 2000 simulations.

The mortality estimates modelled were based on the inverse of survival estimates published graphically in Chilvers and Mackenzie (2010) (with data subsequently clarified by D. MacKenzie) and modified to account for levels of fishing mortality imbedded in them. As these published survival estimates were based on tag re-sights (for age classes 0, 1, 2, 3 and > 4 years) from the Auckland Islands sea lion population from 1998–2005 (Chilvers and Mackenzie 2010), they intrinsically included existing levels of fisheries mortality as well as mortality from epizootics. Therefore, for age classes four years and over (i.e. the age range predominantly killed in fisheries interactions and the non-pup age range reported to die in the 1998 bacterial epizootic) the relevant age-class mortality estimates reported in Chilvers (2012b) were used which had been adjusted to exclude fishing-related and epizootic mortality (Chilvers (2012a); L. Chilvers, personal communication). For age classes less than four years, mortality estimates were averaged from the Chilvers and Mackenzie (2010) data excluding their estimates for 2004 and 2005 which were based on small sample sizes (Table 4.2).

Female sea lions are thought to reproduce from 3–26 years of age (based on evidence of lactation) although most do not breed until they are six years old and no female older than 25 years has been observed with a pup (Childerhouse et al. 2010a; Childerhouse et al. 2010b; MPI 2012). The mean observed reproductive rate for females 3–28 years old was 0.67 (SE = 0.01) (Childerhouse et al. 2010a). Therefore, the '*Age of first offspring for females*' (i.e. age at which the typical female produces her first offspring) was modelled as 6 years, '*Maximum age of reproduction*' for females as 25 years and '*Mean % of adult females producing progeny per year*' as 67% (Table 4.2). It was also assumed that males would continue breeding for at least as long as females (S. Childerhouse, personal communication) and, therefore, '*Maximum age of reproduction*' for males was also modelled as 25 years (Table 4.2).

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

The models published in Hamilton and Baker (2016a) did not include inbreeding depression as it was determined there was limited data available on the population genetics of NZ sea lions, and that inbreeding or strong genetic drift in the Auckland Islands sea lion population was unlikely (Chilvers 2012b; Robertson and Chilvers 2011). However, new research has shown that NZ sea lions from the Auckland Islands population have a moderate level of genetic diversity compared to other pinniped species (Osborne et al., 2016). Therefore, the effect of inbreeding depression on the Auckland Islands population was explored by generating models using the default values recommended for VORTEX (Lacy et al., 2017) (also see Chapter 5, Appendix 5.7.3 (iii)), as well as models without inbreeding depression.

The models published in Hamilton and Baker (2016a) also did not incorporate dispersal as it was considered there was a lack of known movement of breeding females (Chilvers 2012b; Maloney et al. 2009). However, with further consideration, dispersal is now thought to be a likely contributing influence on the Auckland Islands' population and has been factored into the updated models presented in this chapter. Although tagged animal re-sight effort is limited at most sites, based largely on opportunistic re-sights, there is evidence of individual movements (174 males, 45 females, 19 unknown sex) from the Auckland Islands to other locations (see Figure 4.3). While it is acknowledged that the vast majority of these re-sights do not equate to individuals breeding at their new location, possibly due in part to the lack of re-sight effort, there is strong evidence of migration between NZ sea lion sites with new breeding colonies established over the last 30 years. Dispersal from the largest population at the Auckland Islands, which may be at carrying capacity under current environmental conditions (McConkey et al., 2002a; Collins et al., 2016), is likely to have contributed to the population growth on Campbell Island (Childerhouse et al. 2005; Maloney et al. 2012) and the establishment of new, increasing colonies at Stewart Island and the Otago coast (Department of Conservation (DoC) 2018; Roberts and Doonan 2016). Immigration of 10 female and 50 male sea lions was factored into the models of the Campbell Island NZ sea lion population (see Chapter 5, Appendix 5.7.3 (iii)). It is reasonable to assume dispersal (i.e. emigration) from the Auckland Islands population using these same values for emigration to Campbell Island and considering additional dispersal to the Stewart Island and Otago coast populations with a ratio of females to males approximating tag re-sight data (see Chapter 5, Appendix 5.7.3 (iii)). Assuming < 1% annual emigration from the Auckland Island population (based on the 2015 estimate of ~8 000 individuals from Roberts and Doonan (2016)), models included an annual emigration of 70 sea lions (12 female and 58 male) from the Auckland Islands population (Table 4.3), with emigrants distributed across age classes as per the ratios of immigrant sea lions recorded at Otago (McConkey et al., 2002a) (see Chapter 5, Appendix Table A5.1).

The impact of epizootics was modelled through the ‘Catastrophe’ option in VORTEX by assuming a disease event which occurred randomly every 15 years (6.7% frequency) killed half the pups born in that year (Reproduction 0.50) but had no impact on survival of other ages (Survival 1.0). These estimates were based on the highest impact epizootic event recorded at the Auckland Islands in 1997/98 (Chilvers 2008b; DoC and MPI 2016). Although the 1997/98 epizootic also impacted the adult population, this impact was not factored into the modelling because the evidence was that this was not a regular occurrence. There is recent evidence that the frequency and magnitude of disease events may have been underestimated and that disease may be affecting numbers of pups later in the season and, therefore, not detected during the research program (S. Childerhouse, personal communication). Therefore, disease events (affecting half the pup production i.e. Reproduction 0.5) were also modelled at the following hypothetical frequencies: every 4th year (25% frequency), every 2nd year (50% frequency) and every year (100% frequency) (Table 4.2).

As for Chilvers (2012b), it was assumed that animals caught in the Auckland Islands squid fishery, the Auckland Islands scampi fishery and the Auckland Islands non-squid/scampi trawl fisheries were all from the Auckland Islands sea lion population.

Therefore, data from all Auckland Island trawl fisheries were included in the modelling (Table 4.1 and 4.2). Between 2004/05 and 2008/09, after SLEDs were introduced and were being refined, it was reported that females accounted for 71% of the observed number of sea lions captured in the Auckland Islands squid fishery (Robertson and Chilvers 2011). This led to a claim that this bias towards female mortality may be a major contributing factor towards the continued observed population decline at the Auckland Islands as the mortality of breeding females is likely to have a larger impact than male mortality (Robertson and Chilvers 2011). The sex ratio of 77 female:23 male applied in the PVA scenarios (Table 4.2) reflects the observed mean sex ratio of 71 female:29 male scaled to estimated mortalities (MPI 2012). Using the 'Harvest' option in VORTEX, the following different levels of fisheries-related mortality were modelled:

a) As a conservative approach, a 'high' annual loss of 68 adult sea lions (52 females, 16 males) per year was modelled (Table 4.2) based on the 'Fisheries-Related Mortality Limit' set in 2010/11 (MPI 2012). Although SLEDs allow sea lions to escape and survive an interaction with a trawl net, concerns over potential levels of undetected fisheries-related mortality have resulted in a management criterion where the interaction between sea lions and the Auckland Islands squid fishery is managed through a Fisheries-Related Mortality Limit. The Fisheries-Related Mortality Limit has two components: a strike rate (the number of sea lions presumed to be killed in the fishery in the absence of SLEDs, currently set at 5.89 sea lion interactions per 100 trawl hauls based on previous observer data) and a SLED discount rate which provides a discount on this strike rate to reflect the increased likelihood that a sea lion that enters a trawl net will exit via the SLED and survive (currently set at 82%; (MPI 2012));

b) A 'medium' annual mortality level of 28 adult sea lions (22 females:6 males) was modelled based on the average of the maximum 95% confidence level of estimated captures from all Auckland Islands trawl fisheries for the most recent five seasons of reported data (Tables 4.1 and 4.2) (Abraham et al. 2016), and

c) A lower annual mortality level of 13 adult sea lions (10 females, 3 males) per year was modelled based on the average of mean estimated captures from all Auckland Islands trawl fisheries for the most recently reported five seasons (Tables 4.1 and 4.2).

Sensitivity analyses

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

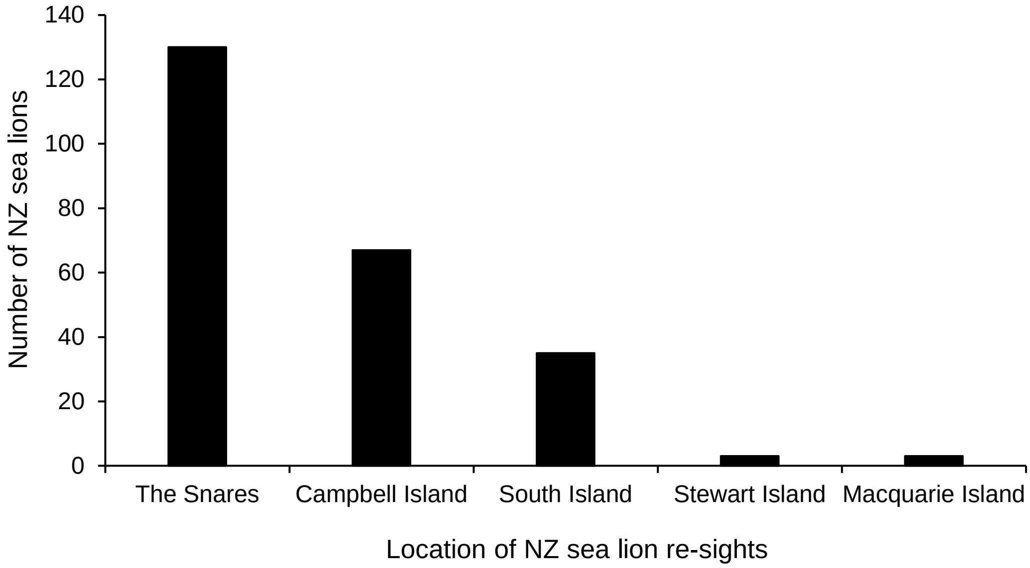


Figure 4.3: For 238 individuals tagged at the Auckland Islands that were re-sighted at a different location, the number seen at each location. Tag re-sight records for 1987–2016 from MPI (2018a).

Table 4.2: NZ sea lion demographic parameters, bycatch levels and mass mortality disease levels used in the Population Viability Assessment (PVA) for the Auckland Islands NZ sea lion population. Further details of how parameter values were derived are in Methods.

Parameter description	Value(s) modelled
Inbreeding Depression	Yes
CV concordance of Reproduction & Survival	None
Breeding strategy	Polygynous (Chilvers 2012b)
Young per year	1
Female breeding age (years)	6 (Chilvers et al. 2010)
Female maximum breeding age (years)	25 (Childerhouse et al. 2010a)
Male breeding age (years)	9 (Robertson et al. 2006))
Male maximum breeding age (years)	25 (S. Childerhouse pers. comm.)
Maximum life span	25
Mean % adult females producing progeny/year (EV = environmental variation)	67% (EV=10) (Childerhouse et al. 2010a)
Sex ratio at birth (males)	51% males (Chilvers 2012b)
Density Dependant Reproduction	No
% males in breeding pool	23 (Robertson et al. 2006)
Female mortality (%)	<4 year mortality data derived from Chilvers and Mackenzie (2010); ≥4 year mortality data from Chilvers (2012b)
Age 0	47 (8)
Age 1	32 (8)
Age 2	20 (6)
Age 3	14 (4)
Age 4	4 (2)
Age 5	4 (2)
Adults	2 (1)
Male mortality (%)	
Age 0	62 (14)
Age 1	34 (8)
Age 2	15 (6)
Age 3	6 (2)
Age 4	2 (1)
Age 5	2 (1)
Age 6	2 (1)
Age 7	2 (1)
Age 8	2 (1)
Adults	2 (1)
Initial population size	12 065 in 2009 (MPI 2012)
Additional scenario options added to Base Models:	
Harvest (Annual bycatch mortality in fisheries)	(i) 68 animals (52F:16M) (ii) 28 animals (22F:6M) (iii) 13 animals (10F:3M)
Catastrophe (Disease)	
- Frequency	(i) 6.7% (ii) 25% (iii) 50% (iv) 100%
- Reproduction	0.50
- Survival	1.0

Table 4.3: NZ sea lion dispersal data from Otago, NZ (columns 1–3) used to estimate the number of emigrant individuals per age class for males and females (columns 5 and 6) in the Population Viability Analysis (PVA) models for the Auckland Islands. Data in column 2 is the age distribution of 40 new identified males (not including pups) sighted from 1995–1999 in Otago from Table 1 in McConkey et al. (2002a). The proportions in column 3, based on the data in column 2, were used to determine the age distributions of males and females in columns 5 and 6.

Age distribution of 40 new identified sightings at Otago from 1996-1999:			Modelled emigration of 70 individuals/year from Auckland Islands:		
Column 1: age	Column 2: # identified	Column 3: proportion	Column 4: age	Column 5: males (M)	Column 6: females (F)
1 year	10	0.25	1–2 years	15	3
2 years	10	0.25	2–3 years	15	3
3 years	3	0.075	3–4 years	4	1
4 years	3	0.075	4–5 years	4	1
5 years	4	0.10	5–6 years	6	1
≥6 years	10	0.25	6–7 yrs (M)/6+ yrs (F)	4	3
Total	**40**	**1.0**	7–8 years	4	–
			8–9 years	3	–
			9+ years	3	–
			Total	**58**	**12**

4.4 Results

The Base Model with emigration (no fisheries-based or epizootic mortality applied) showed NZ sea lion population growth at the Auckland Islands of r = 0.01 and a mean final population size of 17,580 individuals (SD = 3,980) after 30 years of modelling (Model 1, Table 4.4). The Base Model was not affected by the inclusion of inbreeding depression (i.e. growth rate remained at 0.01). Inbreeding depression had no large effect on stochastic r for large or small populations, or if modelled as a population-based model compared to an individual-based model. Therefore, inbreeding depression was excluded from all further models presented in Table 4.4.

Base Model with varying levels of fisheries-related mortality

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

- with an annual mortality reflecting the most recent five years of reported mortality estimates from all Auckland Islands trawl fisheries of 13 (based on annual mean estimated captures) or 28 adult sea lions (based on maximum 95% confidence level of estimated captures), there was no change in the population growth rate (r = 0.01, Model 2 and Model 3 respectively, Table 4.4) compared to the Base Model which had no fisheries-related or epizootic mortality; and,

- at the high Fisheries-Related Mortality Limit of 68 sea lions per year, there was a slight decrease in the population growth rate (r = 0.00, Model 4, Table 4.4) compared to the Base Model.

Base Model with varying frequency of epizootic mortality

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

- every 15 years, there was no substantial decrease in the population growth rate (r = 0.01, Model 5, Table 4.4) compared to the Base Model which had no fisheries-based or epizootic mortality;

- every 4 years, the population growth rate (r = 0.00, Model 6, Table 4.4) was slightly decreased from the Base Model rate;

- every 2 years, there was a negative population growth rate (r = –0.01, Model 7, Table 4.4); and

- every year, showed a negative population growth rate (r = –0.04, Model 8, Table 4.4) resulting in a mean final population size, after 30 years of modelling, that was about one third (N = 3,600, SD = 830, Table 4.4) its original size (N = 12,065).

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

The scenarios with annual epizootic events and additional fisheries-related mortality showed that, with annual mortality of 13 adult sea lions (based on last five years of reported annual mean estimated captures from all Auckland Islands fisheries) or 68 adult sea lions (Fisheries-Related Mortality Limit from MPI (2012)), there was negative population growth rate of r = –0.05 and r = –0.06 (Model 9 and Model 10 respectively, Table 4.4).

All models had a probability of extinction after 30 years of zero, except for Model 10 that showed a probability of extinction of 3.7% (2009–2039).

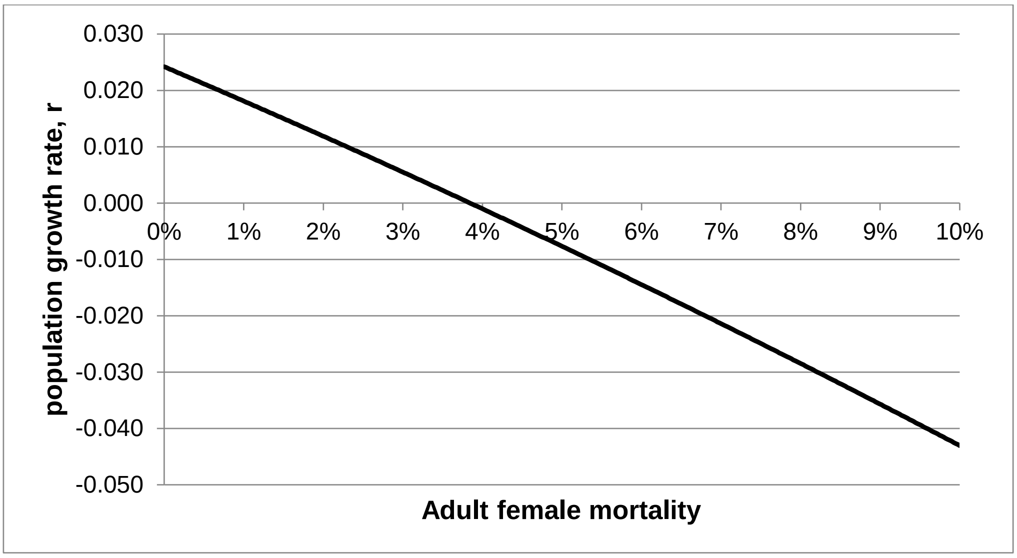
Sensitivity analyses

Sensitivity analyses showed that, in particular, the modelled population was sensitive to annual adult female survival and the proportion of females participating in breeding each year. Whilst female adult survival remained greater or equal to 97%, with no or low bycatch mortality, the population continued to remain stable or with a slight population growth rate (Table 4.4 Models 11 and 12; Figure 4.4). However, with no fisheries mortality applied, adult female survival below 96% resulted in negative population growth (Figure 4.4). In the absence of any fishing mortality, the modelled population continued to grow whilst female breeding participation remained above 58% (Figure 4.4).

Table 4.4: Model predictions for the Auckland Islands NZ sea lion population showing mean stochastic population growth rate (stochastic r), mean final population size (N) and standard deviation (SD) after 30 years. Each model was run for a 30-year period with 2000 simulations.

Model	Description	Model explanation	Mean population change		Mean final population size (to nearest 10)	
			r	SD	N	SD
a	Base Model without emigration	No fisheries-based or epizootic mortality, no emigration	0.01	0.05	18,860	3,980
1	Base Model (with annual emigration of 70)	No fisheries-based or epizootic mortality, annual emigration	0.01	0.05	17,580	3,980
2	Base Model 1 & fishing mortality 13/year at 10F:3M ratio	Low level fisheries-based mortality; no epizootic mortality	0.01	0.05	16,870	3,800
3	Base Model 1 & fishing mortality 28/year at 22F:6M ratio	Medium level fisheries-based mortality; no epizootic mortality	0.01	0.05	16,170	3,820
4	Base Model 1 & fishing mortality 68/year at 52F:16M ratio	High level fisheries-based mortality; no epizootic mortality	0.00	0.05	14,140	3,560
5	Base Model 1 & disease 0.5 reproduction impact (i.e. death of 500F & 500M pups) at 6.7% (every 15 years)	No fisheries-based mortality; epizootic mortality affecting pups every 15 years	0.01	0.06	16,140	3,750
6	Base Model 1 & disease 0.5 reproduction impact (i.e. death of 500F & 500M pups) at 25% (every 4 years)	No fisheries-based mortality; epizootic mortality affecting pups every 4 years	0.00	0.06	13,090	3,300
7	Base Model 1 & disease 0.5 reproduction impact (i.e. death of 500F & 500M pups) at 50% (every 2 years)	No fisheries-based mortality; epizootic mortality affecting pups every 2 years	-0.01	0.06	9,250	2,450
8	Base Model 1 & disease 0.5 reproduction impact (i.e. death of 500F & 500M pups) at 100% (every year)	No fisheries-based mortality; epizootic mortality affecting pups every year	-0.04	0.04	3,600	830

Model	Description	Model explanation	Mean population change		Mean final population size (to nearest 10)	
			r	SD	N	SD
9	Base Model 1 & fishing mortality 13/year at 10F:3M ratio & disease 0.5 reproduction impact (i.e. death of 500F & 500M pups) at 100% (every year)	Low level fisheries-based mortality; epizootic mortality every year	-0.05	0.04	3,280	820
10	Base Model 1 & fishing mortality 68/year at 52F:16M ratio & disease 0.5 reproduction impact (i.e. death of 500F & 500M pups) at 100% (every year)	High level fisheries-based mortality; epizootic mortality every year	-0.06	0.04	2,450	700
	Increasing % annual mortality of adult females					
11	Base Model with adult female mortality = 3%	No fisheries-based or epizootic mortality	0.01	0.05	14,360	3,280
12	Base Model with adult female mortality = 3% & fishing mortality 13/year at 10F:3M ratio	Adult female morality = 3%; low level fisheries-based mortality; no epizootic mortality	0.00	0.05	13,900	3,270
13	Base Model with adult female mortality = 3% & fishing mortality 68/year at 52F:16M ratio	Adult female morality = 3%; high level of fisheries-based mortality; no epizootic mortality	0.00	0.05	11,400	2,910
14	Base Model with adult female mortality = 4%	No fisheries-based or epizootic mortality	0.00	0.05	11,880	2,750
15	Base Model with adult female mortality = 4% & fishing mortality 13/year at 10F:3M ratio	Adult female morality = 4%; low level fisheries-based mortality; no epizootic mortality	0.00	0.05	11,390	2,810
16	Base Model with adult female mortality = 4% & fishing mortality 68/year at 52F:16M ratio	Adult female morality = 4%; high level of fisheries-based mortality; no epizootic mortality	-0.01	0.05	9,080	2,500



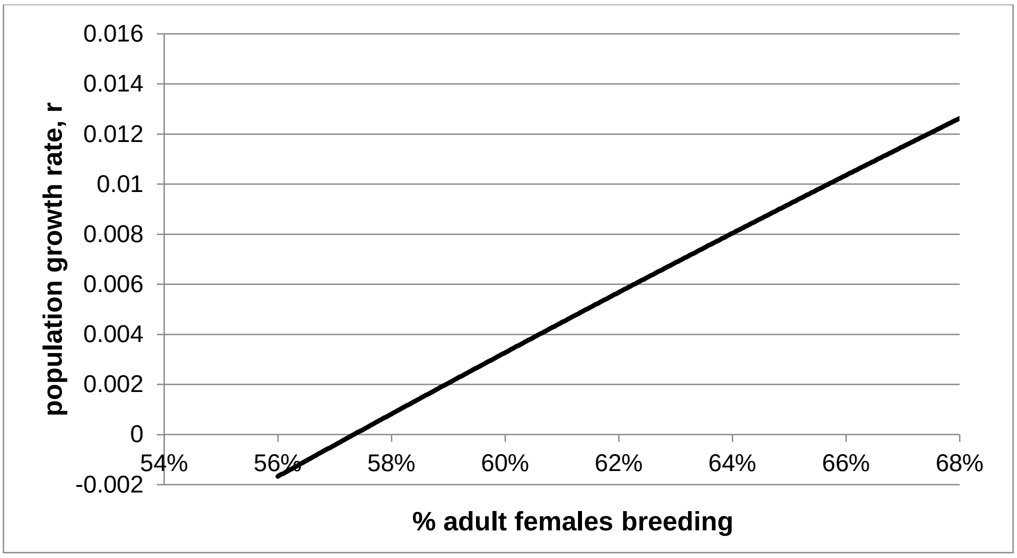


Figure 4.4: Model sensitivity analyses showing the change in NZ sea lion population growth rate given different estimates of adult female annual mortality (top graph) and % female breeding participation (bottom graph). Sensitivity was undertaken using all other parameter values from the Base Model (Table 4.4 Model 1).

4.5 Discussion

New Zealand sea lions from the Auckland Islands population are incidentally killed in trawl fisheries with most of the bycatch having occurred in the Auckland Islands squid fishery. Mitigation management aiming to reduce sea lion bycatch has included the establishment of a 12 nautical mile marine reserve around the Auckland Islands in which all fishing is excluded, the instigation of mortality limits that can trigger spatio-temporal closures, and the design refinement and deployment of SLEDs on all vessels in the Auckland Islands squid fishery since 2004/05 (MPI 2017a). Subsequently, there has been an encouraging reduction in the observed and estimated sea lion bycatch in the Auckland Islands squid fishery (Figure 4.2) (Abraham et al. 2016; Thompson et al. 2013b). Any sea lion that is caught in a trawl net without a SLED will die by drowning whereas correctly deployed SLEDs provide an opportunity for sea lions to escape trawl nets. Extensive efforts to test the efficacy of SLEDs in reducing bycatch have shown that most sea lions are likely to survive following their escape from a trawl net via a SLED (Hamilton and Baker 2015a). To date, SLEDs have not been deployed in any other trawl fishery around the Auckland Islands because of industry concerns about catch loss (Richard Wells, personal communication). However, if sea lion bycatch continues to be reported at the current (or higher) level in the Auckland Islands scampi fishery (Table 4.1), it may be worth investigating the feasibility of deploying SLEDs in this trawl fishery.

This chapter presents updated PVA modelling of the NZ sea lion population at the Auckland Islands. PVA is useful for guiding conservation management and research by identifying the key demographic parameters and impacts that may be affecting the survival of a species. The demographic factors driving trends in the sea lion population were assessed including the impact of different levels of fisheries mortality and epizootic events. The modelling indicated that, even in the absence of both incidental fisheries-related mortality and epizootics, and assuming emigration to other NZ sea lion breeding islands, the population growth rate was low ($r = 0.01$, Model 1, Table 4.4). With no epizootic events incorporated but with the addition of fisheries-related mortality levels reflecting recent levels of reported bycatch (Table 4.1) (Abraham et al. 2016), there was no change in the modelled population growth rate (Models 2 and 3, Table 4.4). There was very little change in the population growth rate even with a relatively high bycatch level of 68 sea lions per year (with 77% female; $r = 0.00$, Model 4, Table 4.4). This indicated that current levels of sea lion bycatch from Auckland Islands trawl fisheries are sustainable, particularly now that effective bycatch mitigation is in place in the Auckland Islands squid fishery. At current levels, fisheries bycatch is unlikely to be the key factor that is driving the population trajectory for this species.

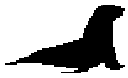
Modelling epizootic events that killed 50% of annual pup production at varying frequencies had a larger impact on the population growth rate than applying bycatch levels. With no fisheries bycatch but with a rate and impact level of epizootic event based on the largest episode recorded in the last 15 years, the population growth rate remained at 0.01 (Model 5, Table 4.4). However, there are recent indications that both the frequency and magnitude of epizootic events may have been underestimated (S. Childerhouse, personal communication). Scenarios modelling more frequent epizootic events in the absence of bycatch effects had a more dramatic impact on population growth rates (e.g. epizootics every 4 years, r = 0.00, Model 7; every 2 years, r = –0.01, Model 7; every year, r = –0.04, Model 8; Table 4.4). Epizootic events, and/or the impact from weather events, affecting pup survival (i.e. 50% reduction in pup production) on an annual basis had the potential to decrease the Auckland Island sea lion population by about 70% over a 30-year time span (Model 8, Table 4.4). The population trajectory from this hypothetical scenario is similar to the decline observed from the mid-1990s to the population low in 2008/09 (Figure 4.1). Modelling both low (13/year, Model 9) and high (68/year, Model 10) levels of bycatch in additional to the high frequency (every year) of epizootic events showed small (0.01) changes in the population growth rate compared to the scenario with no bycatch and high frequency epizootic events. This adds weight to the above conclusion that the current level of bycatch from Auckland Island trawl fisheries is not the main influence on NZ sea lion population growth.

Given the conservative approach taken with the modelling and even with the possibility that a small number of animals could be incidentally caught in other trawl fisheries away from the Auckland Islands, this conclusion is especially likely. The mortality of adult female sea lions is likely to have a larger population impact compared with male mortality. It has been suggested that an apparent increasing female bias in mortality estimates for the Auckland Island squid fishery may have contributed to sea lion population decline (Robertson and Chilvers 2011). Before effective SLEDs were used in the fishery, it was reported that females accounted for 71% of the observed number of sea lions captured (Robertson and Chilvers 2011). However, following SLED refinement, observed capture levels declined to less than 10 individuals a year (Abraham et al. 2016; Thompson et al. 2013b). Hence, any perceived impact of a skewed sex ratio is now unlikely to be significant. Therefore, modelling 77% female bias in capture animals for all Auckland Island trawl fisheries provides a conservative modelling approach, especially when applied to the highest modelled bycatch mortality of 68 animals per year. It should also be noted that this high level does not reflect current mortality data but is a test of the Fisheries-Related Mortality Limit calculated and used by the government fisheries regulator from 2012 to the end of 2017 (MPI 2012; MPI 2017d). Modelling indicated slow population growth when the estimated bycatch levels from all the Auckland Islands trawl fisheries for the past five reported seasons were applied (i.e. 13 or 28 adults, Thompson *et al.,* 2013). However, in the most recent seasons, the reported rates of mortality have been lower than these values. For example, in 2012/13, for Auckland Islands squid, scampi and non-squid/scampi fisheries combined, the mean estimated mortality was 10 animals (Table 4.1; Thompson *et al.,* 2013), and there were no NZ sea lions observed killed in the most recent (2019/20) fishing season with almost 100% observer coverage (Fisheries NZ, unpubl. data).

Sensitivity analysis can help measure the relative influences of different demographic parameters on population predictions. In particular, the modelling of the Auckland Islands sea lion population was sensitive to changes in adult female annual mortality and the proportion of adult females that breed each year. After adjusting each parameter separately, and in the absence of bycatch or epizootic mortality, population growth rate was negative once adult female mortality fell below 4%, and also when the proportion of females breeding each year fell below 58% (Figure 4.4), indicating that the continued NZ sea lion population decline at the Auckland Islands may have perhaps been related to factors affecting decreased breeding productivity.

These PVA modelling results are consistent with other recent NZ sea lion population modelling which showed poor correlations between survival of juveniles (ages 2–5 years) and adults (6–14 years) and fishery-related mortality in the Auckland Islands squid fishery indicating that variation in vulnerable age classes was not primarily driven by the direct effects of fishing (Roberts and Doonan 2014; Roberts and Doonan 2016). With continued commitment to mitigation and maintaining adequate observer coverage in the trawl fisheries around the Auckland Islands, further reductions in estimated mortality levels should be achievable. If direct impacts of fishing operations are no longer a significant problem, resources should be directed towards determining other hypotheses to explain any further suppression of the sea lion population trajectory. From the modelling presented here, the severity and frequency of epizootic events and their effect on annual pup production provides a more plausible explanation for the NZ sea lion population decline observed at the Auckland Islands from the mid-1990s. Over the past decade, the population trend of NZ sea lions at the Auckland Islands has been, at the least, stable with inter-annual variability in pup counts (Dodge et al. 2019; Melidonis and Childerhouse 2020). Focus on management options that can increase productivity, such as mitigation of disease (Michael et al. 2019) and other factors affecting pup survival, warrants priority.

Chapter 5 — Population growth of an endangered pinniped – the New Zealand sea lion (*Phocarctos hookeri*) – is limited more by high pup mortality than fisheries bycatch



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5.1 Abstract

The endangered New Zealand sea lion, *Phocarctos hookeri* is killed as incidental bycatch in a trawl fishery operating near their second largest population on Campbell Island in New Zealand's sub-Antarctic. Using the Potential Biological Removal (PBR) procedure to assess the sustainability of this bycatch for the sea lion population on Campbell Island indicated that annual bycatch estimates, particularly following the implementation of bycatch mitigation measures, are below the PBR threshold of 25 (derived using a precautionary approach). Preliminary Population Viability Analysis (PVA) modelling supported the finding that current bycatch levels, especially given a strong male bias (98%) in bycatch, are sustainable for this population. Models showed that reducing pup mortality through management actions, such as installing ramps in wallows where large numbers of pups drown, would lead to increased population growth. While obtaining more accurate data on population status and demographic parameters for the Campbell Island population should be a priority, this will take many years of research. The PBR and PVA tools demonstrate that contemporary conservation management should continue to focus on increasing pup survival while maintaining mitigation approaches that have reduced bycatch to low levels, together with high observer coverage to sustain confidence in annual bycatch estimates.

5.2 Introduction

The endemic and endangered New Zealand (NZ) sea lion (*Phocarctos hookeri*, rāpoka) (Chilvers 2015), once bred from the northernmost cape of the North Island to the sub-Antarctic islands of NZ (Childerhouse and Gales 1998). Historically, commercial harvesting reduced the overall population size and breeding distribution (Childerhouse et al. 2010b), leaving breeding populations now concentrated at the Auckland Islands (two main areas), Campbell Island, Stewart Island and the Otago coast of the South Island (DoC and MPI 2016) (see Figure 3.1). Based on 2018 (i.e. the 2017/18 breeding season) pup production estimates (Childerhouse et al. (2018); NZ Department of Conservation, unpublished data), 69% and 29% of all NZ sea lion pups are born at the Auckland Islands and Campbell Island, respectively. Over the past 30–40 years, while there has been overall population decline with a stabilization of numbers in more recent years, there have been differences in the population trajectories at each breeding location. The Auckland Islands population declined by about half from the late–1990s to late–2000s, with numbers stabilizing over the past decade; the Campbell Island population has undergone rapid population growth; and new populations have established and are slowly increasing on Stewart Island and the Otago coast (Childerhouse et al. 2018; Maloney et al. 2012; Roberts and Doonan 2016).

The potential threats to NZ sea lions include fisheries bycatch, disease-related mortality, food limitations (either due to climate or fisheries influences), deliberate human-induced mortality, poor breeding habitat quality and direct predation (e.g. dogs, sharks), with each breeding population vulnerable to a different suite of threats (Roberts and Doonan 2016). The NZ mainland (Otago coast) population is mainly susceptible to deliberate human-induced mortality, predation from dogs and mortality in set nets (gillnets); the Stewart Island population to direct and indirect fisheries impacts and deliberate human-induced mortality; the Auckland Islands population to fisheries bycatch, disease-related mortality and pup mortality; and the Campbell Island population to fisheries bycatch and pup mortality due to poor quality breeding sites (DoC and MPI 2016; Roberts and Doonan 2016). Another potential influence on population dynamics and trajectory is the dispersal of individual sea lions. Previous population modelling was based on the assumption that dispersal between colonies was negligible as NZ sea lions, particularly adult females, are considered to be philopatric (Chilvers 2012b; Chilvers and Meyer 2017; Chilvers and Wilkinson 2008; Hamilton and Baker 2016a). However, there is evidence of dispersal from tagged animals (Geschke and Chilvers 2009; MPI 2018a; Robertson et al. 2006), the lack of population structure between breeding colonies based on mitochondrial DNA and nuclear loci analyses (Collins et al. 2017), and, unequivocally, the establishment of new breeding colonies along the Otago coast and Stewart Island (Chilvers 2018; McConkey et al. 2002a; McConkey et al. 2002b).

The NZ sea lion population on Campbell Island has been increasing following their presumed near extirpation by harvesting in the nineteenth and early twentieth centuries (Childerhouse et al. 2005). There are anecdotal records of up to 20 females in 1947 (Bailey and Sorensen 1962) and small numbers of pups in the 1960s (Childerhouse and Gales 1998). Although survey timing and methodology were less consistent prior to 2008, total pup production estimates have increased to 734 pups in 2018 (Childerhouse et al. (2015); NZ Department of Conservation, unpublished data) (Figure 5.2). The particularly high levels of pup mortality (~40–60%) at this location have been mainly attributed to drowning and trauma from falling into natural holes (also known as ‘terrain traps’), adult male aggression and disease (Childerhouse et al. 2015; DoC and MPI 2017; Roe et al. 2015). The only fishery around Campbell Island which has recorded NZ sea lion bycatch is the southern blue whiting (*Micromesistius australis*) trawl fishery, hereafter referred to as the SBW fishery, operating in >400 m depths between August and November each year (Abraham and Berkenbusch 2017; Thompson et al. 2015). Except for one or two females, bycatch has mainly comprised male sea lions, with the first observed capture in 2002 and bycatch remaining low over the subsequent decade (Abraham and Berkenbusch 2017) (Figure 5.3). The fishery was certified as sustainable by the Marine Stewardship Council (MSC) in 2012 (MSC 2018), with a range of bycatch mitigation measures implemented including offal management protocols and the cessation of trawling in the vicinity of large numbers of sea lions (Deepwater Group 2017a). A 2013 peak in bycatch of 21 males led to an “expedited audit”, as per MSC Certification Requirements, resulting in all vessels installing Sea Lion Exclusion Devices (SLEDs) (Hamilton and Baker 2015a) and employing 100% observer coverage (Deepwater Group 2017b; MSC 2018) (Figure 5.3).

In contrast, the Auckland Islands’ population suffered a substantial decline for over a decade (Childerhouse et al. 2018) attributed to a combination of impacts, particularly disease events (DoC and MPI 2016; Roberts and Doonan 2016) and bycatch of mostly adult females in Auckland Islands’ trawl fisheries (Abraham and Berkenbusch 2017; Hamilton and Baker 2016a). NZ sea lion tracking data have shown differential at-sea foraging distributions for males and females (Chilvers 2008a; Chilvers 2009; Geschke and Chilvers 2009), with female-skewed bycatch in Auckland Islands’ trawl fisheries owing to the foraging areas of females from the Auckland Islands overlapping with, in particular, the Auckland Islands squid trawl fishery (Chilvers 2008a; Chilvers 2009), and the male-skewed bycatch in the SBW fishery due to the foraging areas of males from Campbell Island overlapping with this fishery (M-A Lea, unpublished data).

The conservation management of the Auckland Islands' population has been controversial and highly debated, with a range of modelling undertaken to assess impacts on this population (Breen et al. 2012; Chilvers 2012a; Chilvers 2012b; Hamilton and Baker 2016a; Hamilton and Baker 2016b; Meyer et al. 2015a; Meyer et al. 2015b; Meyer et al. 2016; Meyer et al. 2017; Meyer et al. 2018; Middleton and Breen 2016; Roberts et al. 2018; Roberts and Doonan 2016; Roberts et al. 2014a; Robertson 2015). While there has been focussed attention on the role that fisheries bycatch impacted the Auckland Islands' population, there is agreement that no single factor has been solely responsible for this population's decline (DoC and MPI 2017; Hamilton and Baker 2016a; Roberts and Doonan 2016) and, furthermore, bycatch has been effectively mitigated (DoC and MPI 2017; Hamilton and Baker 2015a; Hamilton and Baker 2016a; Roberts et al. 2018). These debates have not been replicated and no population models have been previously published for the Campbell Island population. This is presumably due to less concern regarding the lower fisheries bycatch levels, the male-biased bycatch in the SBW fishery (compared to female-biased bycatch at the Auckland Islands at a time when females are feeding pups) (Thompson et al. 2013b), and the increasing trajectory of this population. The differences between the two populations warrant a focussed assessment of the factors affecting the Campbell Island population.

A widely used approach to assess bycatch sustainability is calculation of a Potential Biological Removal (PBR) level, which is the maximum amount of non-natural mortalities a population can incur while staying above half its carrying capacity in the long term (Wade 1998). While PBR assessments for the Campbell Island population were carried out in 2012 and 2014 as part of the MSC process (MSC 2018), these were prior to the up-listing of NZ sea lions to an endangered status (Chilvers 2015), and the most recent on-island pup counts which indicate a potential slowing in growth for this population (Childerhouse et al. (2015); NZ Department of Conservation, unpublished data) (Figure 5.2). Population Viability Analysis (PVA) supports and advances the PBR approach by providing a tool for assessing bycatch impact, particularly when bycatch sex ratios are uneven, as well as determining key demographic factors driving population trajectory. In the absence of much population-specific demographic data for NZ sea lions at Campbell Island, a preliminary population assessment, particularly the relative efficacy of different management actions, can be appropriately achieved using modelling packages such as VORTEX (Hamilton and Moller 1995) and, as a proxy, published demographic data from long-term sea lion monitoring at the Auckland Islands (Childerhouse et al. 2018; Childerhouse et al. 2010a; Childerhouse et al. 2010b; Chilvers 2012b; Chilvers and Mackenzie 2010). The modelling packages simulate survival and reproductive events in successive years for each individual in a population by the Monte Carlo method and are stochastic in that they impose variations in annual survival and reproduction by random number generations according to prescribed probability distributions for reproduction and survival rates (Lacy et al. 2017). These packages are subject to wide scrutiny; repeatedly used, frequently revised and updated; easy-to-understand, replicate and verify; and have been widely applied to assist research and management of threatened species (Brook et al. 2000; Hamilton and Moller 1995; Lacy 2019).

In this context, the aims of this chapter are to a) determine PBR levels for the Campbell Island NZ sea lion population; and b) undertake a preliminary PVA to further evaluate the impact of fisheries bycatch on this population, particularly considering the male bias in bycatch and the impact of intervention to reduce pup mortalities both in the absence and presence of bycatch.

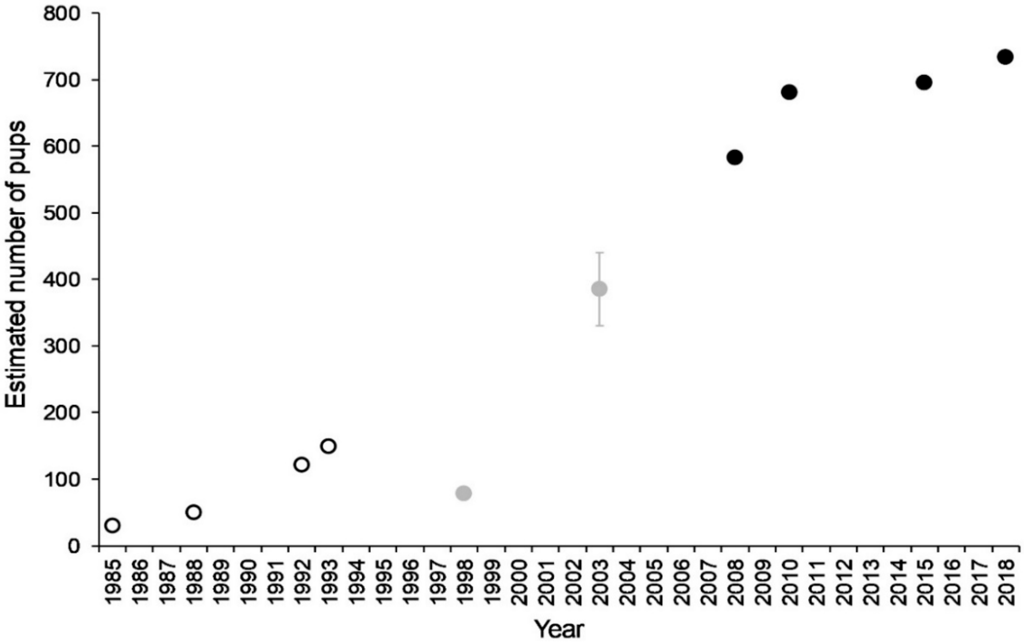

Figure 5.1:NZ sea lion pup count estimates for Campbell Island from 1985–2018. Empty Open circles indicate direct pup counts; dark grey circles indicate a mixture of mark-recapture and direct counts (noting that 1998 was an incomplete count, McNally et al. (2001)) with 95% confidence intervals from the mark-recapture component in 2003; and, from 2008 onwards, black circles indicate estimates using consistent mark-recapture methodology based on counts of tagged pups and untagged dead pups. Data from Childerhouse et al. (2005) (and references within), Childerhouse et al. (2015) and NZ Department of Conservation (unpublished data).

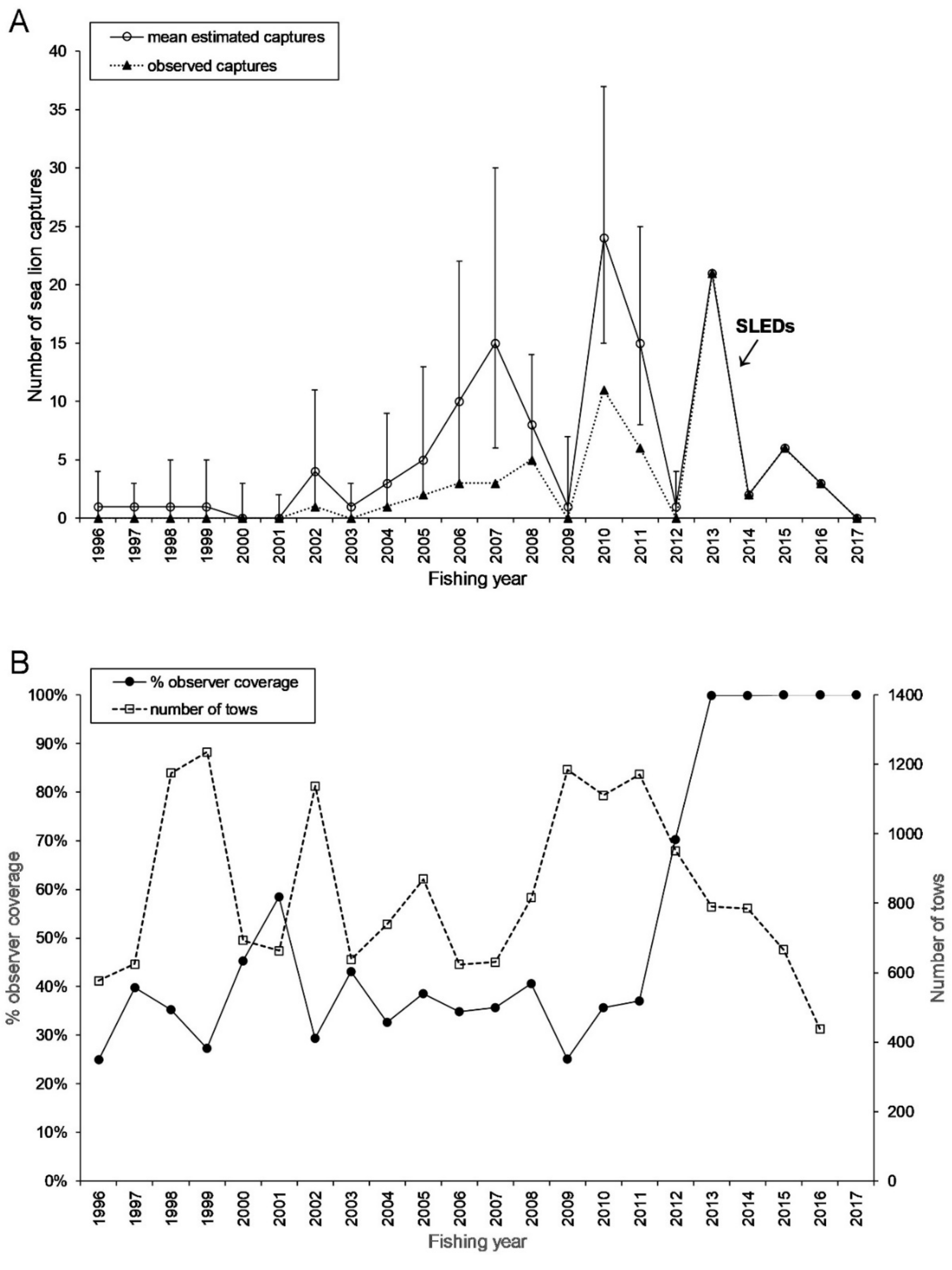

Figure 5.2: From 1996–2017, A) the annual observed number of captures (black triangles) and mean estimated captures (black open circles; error bars = 95% confidence interval) of NZ sea lions, and B) % observer coverage (black circles) and number of tows (open squares) for the Campbell Island SBW fishery. The red arrow indicates when Sea Lion Exclusion Devices (SLEDs) were deployed in this fishery following the 2013 season. Data from 1996–2016 from MPI (2018b), Abraham and Berkenbusch (2017) and, for 2017, NZ Ministry for Primary Industries (unpublished data).

5.3 Methods

Population size estimate for Campbell Island

Pinniped breeding population sizes, particularly those with limited data, are commonly estimated using pup counts multiplied by a value which is the ratio of the total population to the annual number of pups produced (Shaughnessy and McKeown 2002). In the absence of population-specific demographic data for Campbell Island and applying a precautionary approach, a pup multiplier of 4.5 was selected (see Appendix 5.A). Applied to the estimate of 734 NZ sea lion pups in 2018 (NZ Department of Conservation, unpublished data), a population size of 3 300 (rounded to nearest 10) was derived for the PBR calculations, as well as the PVA models (Tables 5.1 and 5.2).

Potential Biological Removal (PBR)

PBR estimates were calculated for the Campbell Island NZ sea lion population following Wade (1998), using:

$$PBR = N_{\min} \tfrac{1}{2} R_{\max} F_r$$

where

N_{min} is the *minimum population* estimate of the population being assessed. While the Campbell Island population estimate of 3 300 is based on the minimum pup number (734) in 2018, the selection of a multiplier (4.5) is more arbitrary. To incorporate this uncertainty, N_{min} was calculated as recommended by Wade (1998) giving an estimate of 2 793 (see Appendix 5.B (i) for more details);

$\frac{1}{2} R_{max}$ is one-half the maximum theoretical or estimated *net productivity rate* of the population at a small population size. Based on theoretical productivity rates fitted to pup count estimates from Campbell Island over a 34-year period (see Appendix 5B (ii) and Figure 5B.1), an R_{max} of 0.06 was considered a reasonable fit and selected as the base value in PBR calculations. R_{max} values of 0.04 and 0.08 were also used as an indicator of the sensitivity of PBR levels if the population growth rate decreased or increased; and

F_r is a *recovery factor* between 0.1 and 1, which acts as a 'safety' parameter to account for unknown biases in estimates. As the Campbell Island population is above the recommended critical abundance threshold of 1 500 and has an increasing trajectory, an F_r value of 0.3 was considered most appropriate

(Moore and Merrick 2011; Taylor et al. 2003) (see Appendix 5B (iii)). As an indicator of the sensitivity of PBR to changes in F_r, PBR estimates were also calculated for F_r of 0.2 and 0.4.

Population Viability Analysis (PVA) modelling

A population model for NZ sea lions at Campbell Island was developed using VORTEX, Version 10.2.17.0 (Lacy and Pollak 2017). As the focus was on assessing population- and fisheries-specific impacts, a single population PVA was undertaken rather than a meta-population assessment. The population dynamics of NZ sea lions have been studied at both the Otago coast and Auckland Islands. Demographic characteristics vary significantly between these two populations (Augé et al. 2011; Augé et al. 2012; Chilvers et al. 2006). It was considered the Campbell Island population may be demographically more similar to the Auckland Islands than the Otago population, based on foraging ecology data (M-A Lea, unpublished data), and their proximity in the NZ sub-Antarctic. Furthermore, the Otago population is considered atypical, partly as it originated from a single matriarch (Augé et al. 2011; MPI 2017a). Therefore, the modelling approach followed Hamilton and Baker (2016a) and was based on the most robust population dynamics data available for NZ sea lions from the Auckland Islands population (input parameters summarised in Table 5.1). Where available, parameter values specific to Campbell Island are described below, with further details on input values using proxy data from the Auckland Islands population provided in the Appendices and Hamilton and Baker (2016a). Each model was run for a 30-year period with 2 000 simulations, a carrying capacity of 10 000 individuals and with 'quasi-extinction' set at a final population of <1 000 individuals.

Inbreeding depression. NZ sea lions from the Auckland Islands population have a moderate level of genetic diversity compared to other pinniped species (Osborne et al. 2016). The effect of inbreeding depression on the Campbell Island population was explored by generating models using the default values recommended for VORTEX (Lacy et al. 2017) (see Appendix 5C (iii)), as well as models without inbreeding depression.

Mortality rates. As pup mortality data were available for Campbell Island, this was used in the estimate for mortality in the first year. A mortality rate of 0.59 was calculated for males and females in their first year (Age 0, Table 5.1) using an average mortality of 0.47 in the first two months following birth based on five pup mortality estimates (range 0.36–0.47) from Campbell Island (Childerhouse et al. 2015; Childerhouse et al. 2005; Maloney et al. 2009; Maloney et al. 2012; McNally et al. 2001), with an additional 0.12

mortality applied for the remainder of the first year derived from Auckland Islands data (Roberts and Doonan 2016). Note that this is likely to be an underestimate as research teams have opportunistically extracted pups from inescapable mires (Childerhouse et al. 2015; Maloney et al. 2009; Maloney et al. 2012). Due to increasing efforts to stop pups drowning in mud wallows in 2018 (S. Childerhouse, pers. comm.), pup mortality data from that year were not included.

Sub-adult and adult mortality rates from the Auckland Islands population were used as a proxy in the Campbell Island models. Mortality rates for NZ sea lions have been generated using different approaches and different subsets of data from the well-studied Sandy Bay (Auckland Islands) colony (Chilvers and Mackenzie 2010; Meyer et al. 2015b; Middleton and Breen 2016; Roberts et al. 2014a). Mortality values generated by Chilvers and Mackenzie (2010) were based on the full tag re-sight database and were adjusted to remove estimated mortality attributed to fisheries bycatch and epizootics (Chilvers 2012a; Chilvers 2012b), which enabled modelling of a range of fisheries and disease impact scenarios for the Auckland Islands population (Chilvers 2012b; Hamilton and Baker 2016a). Therefore, these sub-adult and adult mortality estimates were also considered appropriate for the Campbell Island models (Table 5.1; also see Appendix 5C (ii)).

Immigration. Immigration was factored into the models as a likely contributing influence on the recovery and growth of the Campbell Island population, with the source of dispersal probably from the largest population at the Auckland Islands which, under current environmental conditions, may be at carrying capacity (Collins et al. 2016; McConkey et al. 2002a).

From 1995–1999, 113 migrant males and four migrant females were identified from the NZ sea lion population on the Otago coast (McConkey et al. 2002a), giving an average rate of 23 new arrivals per year, with 3–4% being female. Over this period, while most migrant males were ≤ 2 years old at arrival, there was a spread across age classes (McConkey et al. 2002a) (Appendix 5C (iii) and Table 5C.1).

Tag re-sight data from 1987–2016 (MPI 2018a) provided further indication of the degree of movement of males and females between locations (Appendix 5C (iii) and Figure 5C.1). Of 456 known-sex, tagged individuals re-sighted from 1987–2016 at a different location from where they were tagged (MPI 2018a), 79% (360) were male and 21% (96) were female. Of 73 individuals tagged on the Auckland Islands and re-sighted at Campbell Island, 15% (11) were female and 85% (62) were males (Table 5C.2 in

Appendix 5C). While there are few records of individuals breeding at their new location, this may be partly due to limited search effort. It is reasonable to assume immigration into the Campbell Island population with a ratio of females to males approximating the tag re-sight ratio (Table 5C.1). Assuming < 1% annual emigration from the Auckland Island population (based on the 2015 estimate of ~8 000 individuals from Roberts and Doonan (2016)), models included an annual immigration of 60 NZ sea lions (10 females and 50 males) into the Campbell Island population (Table 5.1), with immigrants distributed across age classes as per the ratios of immigrant sea lions recorded at Otago (McConkey et al. 2002a) (Table 5C.1 in Appendix 5C).

Assessing influence of reduced pup mortality

As a large proportion of NZ sea lion pup mortality has been attributed to holes, drops or barriers within breeding sites that cause pups to drown or be separated from their mothers (MPI 2017a), installation of wooden ramps in mud wallows and, potentially, fencing 'dangerous' sites are identified as management measures to increase pup survival (DoC and MPI 2017). For five seasons, the average total pup mortality rate at Campbell Island was 0.47 (see above), and an average of 0.42 of pup deaths were attributed to trauma-related incidents (Roberts and Doonan 2016). To simulate measures to reduce pup mortality, PVA modelling included scenarios where trauma-related pup mortality was reduced by half using:

$$R = P - [\tfrac{1}{2}TP]$$

where R is the reduced pup mortality (with 'trauma-related' deaths halved); P is the original pup mortality; and T is the proportion of pup deaths that were 'trauma-related'. Therefore, the reduced pup mortality, R, was calculated as 0.37. Applying additional mortality of 0.12 for the remainder of the year (as above) resulted in a reduced first year mortality of 0.49 for both males and females. Scenarios with reduced first year mortality were modelled in the absence and presence of fisheries bycatch.

Assessing impact of fisheries bycatch

It was assumed that NZ sea lions caught in the SBW fishery originated from the Campbell Island population (note that two males caught in this fishery were tagged as pups on Campbell Island, Thompson et al. (2015)), and all bycatch individuals were either dead or died subsequent to capture, although a small number were released alive (Thompson et al. 2015). To date, there are no reports of tagged Auckland Islands' NZ sea lions being killed in the SBW fishery (NZ Department of Conservation., unpublished data), although some males from the Auckland Islands forage around Campbell Island and may spatially overlap with this fishery (Geschke and Chilvers 2009).

From 1996–2005, the mean estimated NZ sea lion captures in the SBW fishery was < 6 per annum, with bycatch increasing from 2006–2013 (prior to SLED use), although annual estimated captures were highly variable (Abraham and Berkenbusch 2017) (Figure 5.3). Fishing effort (number of tows) has also varied with no obvious correspondence between increases in effort and increased sea lion captures (Figure 5.3). For five years prior to SLED implementation (2009–2013), the average of annual mean estimated captures was 12 (range 1–24), and for four years of available data since SLED implementation (2014–2017), the average was 3 sea lions (range 0–6) (Abraham and Berkenbusch 2017; MPI 2018b). With 100% observer coverage since 2013, estimated captures are equivalent to observed captures.

PBR assessment. A standard assumption under the PBR model is that there is no age or breeding colony bias in the distribution of bycatch animals (Wade 1998). Guidelines specify that F_r can be adjusted to accommodate additional information and to allow for management discretion as appropriate (Moore and Merrick 2011). To account for the 98% male bias in SBW fishery bycatch (Abraham et al. 2016; Thompson et al. 2015), an additional PBR estimate was calculated using an F_r of 0.6 (i.e. double the base value of 0.3) (Table 5.2). The bycatch levels in the SBW fishery were compared with PBR estimates to evaluate bycatch sustainability.

PVA models. Due to the strong male bias (98%) in bycatch, the following scenarios were modelled:

- ‘High’ bycatch: the highest annual mean estimated capture, prior to SLED implementation, of 24 NZ sea lions in 2010 (Abraham and Berkenbusch 2017), applied as an annual ‘harvest’ of 23 adult males and one adult female;
- ‘Medium’ bycatch: the average of five years of annual mean estimated captures prior to SLED implementation (2009–2013, see above), applied as an annual ‘harvest’ of 11 adult males and one adult female; and,
- ‘Low’ bycatch: the average of four years of annual mean estimated captures since SLEDs were implemented (2014–2017, see above), applied as an annual ‘harvest’ of two adult males and one adult female (Table 5.1).

To explore the hypothetical impact if fisheries bycatch included more adult female NZ sea lions, scenarios with ‘high’ bycatch comprising 50% and 100% adult females were also modelled. PVA analysis was also used to test the sustainability of calculated PBR levels.

Table 5.1: NZ sea lion demographic parameters and bycatch levels used in the Population Viability Analysis (PVA) for the Campbell Island NZ sea lion population.

Parameter description	Value(s) modelled
Inbreeding Depression*	Yes
CV concordance of Reproduction & Survival	None
Breeding strategy	Polygynous (Chilvers 2012b)
Young per year	1
Female breeding age (years)	6 (Chilvers 2012b; Chilvers et al. 2010)
Female maximum breeding age (years)	25 (Childerhouse et al. 2010a)
Male breeding age (years)	9 (Robertson et al. 2006)
Male maximum breeding age (years)	25 (S. Childerhouse pers. comm.)
Maximum life span	25
Mean % adult females producing progeny/year (EV = environmental variation)	67% (EV=10) (Childerhouse et al. 2010a)
Sex ratio at birth (males)	51% males (Chilvers 2012b)
Density dependant reproduction	No
% males in breeding pool	23 (Robertson et al. 2006)
Female mortality (%)*	Age 0 mortality from Campbell Island population data; Age 1-3 mortality data derived from (Chilvers and Mackenzie 2010); ≥ Age 4 mortality data from (Chilvers 2012b)
Age 0	59 (8)
Age 1	32 (8)
Age 2	20 (6)
Age 3	14 (4)
Age 4	4 (2)
Age 5	4 (2)
Adults	2 (1)
Male mortality (%)*	
Age 0	59 (14)
Age 1	34 (8)
Age 2	15 (6)
Age 3	6 (2)
Age 4	2 (1)

Parameter description	Value(s) modelled
Age 5	2 (1)
Age 6	2 (1)
Age 7	2 (1)
Age 8	2 (1)
Adults	2 (1)
Initial population size*	3 300
Immigration (modelled as 'Supplementation')*	50 males and 10 females per year
Bycatch scenarios:	
Removal of bycatch individuals ('Harvest' model option)*	(i) "High" = 24 adult sea lions per year (1 female & 23 males)
	(ii) "Medium" = 12 adult sea lions per year (1 female & 11 males)
	(iii) "Low" = 3 adult sea lions per year (1 female & 2 males)
Adjusted pup mortality scenarios:	% female mortality in first year (Age 0): 49 (8)
i.e. pup mortality attributed to 'trauma' reduced by half*	% male mortality in first year (Age 0): 49 (14)

Derivation of parameter values are described in Hamilton and Baker (2016a) and in the text.

*see text for sources of values

5.4 Results

Potential Biological Removal (PBR)

Using a minimum population size of 2 793 and the selected values of 0.06 for R_{max} and 0.3 for F_r, a PBR estimate of 25 was calculated for the Campbell Island NZ sea lion population (Table 5.2). As PBR estimates assume an equal sex ratio of bycatch individuals, this equates to approximately 12 female and 12 male NZ sea lions per year. Increasing F_r to 0.6 to account for strong male bias in bycatch numbers resulted in a PBR of 50. Lowering either the productivity rate (R_{max}) or the recovery factor (F_r) resulted in lower PBRs and, conversely, increasing these values produced higher PBR estimates (Table 5.2). As PBR is directly proportional to population size, doubling population numbers, for example, would double the PBR estimate.

Population Viability Analysis (PVA)

When developing base models using known demographic parameters for NZ sea lions (Table 5.1), results indicated that the Campbell Island population would decline in the absence of immigration (Base Model 2), which does not fit the known population trajectory (Figure 5.2). Based on evidence of movement between colonies and immigration to establish new colonies (McConkey et al. 2002a; McConkey et al. 2002b), Base Model 1 included a low level of immigration into the Campbell Island population (Appendix 5C Tables 5C.1 and 5C.2). There was a reasonable correspondence between the predicted trajectory (using Base Model 1 parameters) and population estimates (using observed pup counts) for 2008–2018 at Campbell Island, whereas the predicted population trajectory without immigration did not correspond well with population estimates (Figure 5.4).

Base Model 1, using survival parameters from the Auckland Islands' NZ sea lion population adjusted to exclude levels of fisheries mortality and epizootic events (Chilvers 2012a; Chilvers 2012b; Hamilton and Baker 2016a) and with higher pup mortality (Campbell Island data), did not show population decline after a projected 30 years, although population growth was low (Table 5.3). Modelling annual bycatch mortality of up to 24 sea lions, with a strong male bias, did not lead to population decline: applying bycatch reflecting the average estimated capture levels since SLEDs were implemented (i.e. 3 sea lions/year; Model 3) indicated no change from the Base Model 1 population growth rate, although there may be slightly lower growth rates with annual bycatch of 12 or 24 individuals (Models 4 and 5, Table 5.3). An annual bycatch of 50 sea lions with 98% male bias indicated the population would still maintain growth (Model 7, Table 5.3). However, when scenarios included a higher proportion of adult females in bycatch numbers, the impact may be greater (Models 5.1 and 5.2, Table 5.3), and annual bycatch of 34 sea lions with an equal sex ratio indicated zero population growth (Model 6, Table 5.3).

Simulating a management scenario where pup mortality attributed to 'trauma' was reduced by half indicated a doubling of the Base Model 1 population growth rate to 0.021 (Model 8). Applying 'low' bycatch to this model showed no change to population growth (Model 9) and applying 'medium' (Model 10) and 'high' bycatch (Model 11) indicated slightly lower population growth (Table 5.3).

None of the models resulted in 'quasi-extinction' (i.e. <1 000 individuals) of the population over the next 30 years. While there were differences in the mean final population estimates and mean population growth rates derived under various bycatch and pup mortality mitigation scenarios, the large, overlapping standard deviations (Table 5.3) indicated the differences were not statistically significant. The inclusion of inbreeding depression in the models did not alter population growth rates.

Table 5.2: Potential Biological Removal estimates (PBRs) for the New Zealand sea lion population on Campbell Island using a minimum population size of 2 793 and a range of recovery factor (F_r,) and maximum theoretical net productivity rate (R_{max}) values.

	R_{max} 0.04	R_{max} 0.06	R_{max} 0.08
F_r = 0.2	11	17	28
F_r = 0.3	17	**25**	34
F_r = 0.4	22	34	45
F_r = 0.6	34	50	67

PBR highlighted in bold represents the estimate derived using the most appropriate parameter values (see text for details). Other PBR estimates are presented to show sensitivity of PBR to variations in parameters.

Table 5.3: Model predictions for the NZ sea lion population on Campbell Island showing mean stochastic population growth rate (r, with standard deviation, SD; estimates rounded to 3 decimal places) and mean final population size (N, with SD; estimates rounded to nearest 10) after 30 years.

Model	Description	Mean population change		Mean final population size	
		r	SD	N	SD
1	Base Model (which includes immigration)	0.009	0.044	4 380	750
2	Base Model with no immigration	-0.003	0.048	3 080	660
3	Base Model 1 with 'low' bycatch mortality 3/year (1 adult female:2 adult males)	0.009	0.044	4 320	750
4	Base Model 1 with 'medium' bycatch mortality 12/year (1 adult female:11 adult males)	0.008	0.044	4 230	730
5	Base Model 1 with 'high' bycatch mortality 24/year (1 adult female:23 adult males)	0.007	0.045	4 110	740
5.1	Base Model 1 with 'high' bycatch mortality 24/year (12 adult females:12 adult males)	0.003	0.043	3 670	660
5.2	Base Model 1 with 'high' bycatch mortality 24/year (all adult female)	-0.002	0.041	3 140	570
6	Base Model 1 with bycatch mortality 34/year (17 adult females:17 adult males)	0.000	0.043	3 360	630
7	Base Model 1 with bycatch mortality 50/year (1 adult female:49 adult males)	0.005	0.047	3 860	750
8	Base Model 1 with % pup mortality attributed to 'trauma' decreased by half & no bycatch mortality	0.021	0.045	6 290	1 080
9	Base Model 1 with % pup mortality attributed to 'trauma' decreased by half & 'low' bycatch mortality (3/year)	0.021	0.045	6 200	1 060
10	Base Model 1 with % pup mortality attributed to 'trauma' decreased by half & 'medium' bycatch mortality (12/year)	0.020	0.045	6 080	1 080
11	Base Model 1 with % pup mortality attributed to 'trauma' decreased by half & 'high' bycatch mortality (24/year)	0.019	0.046	5 980	1 090

Each model was run for a 30-year period with 2 000 simulations, with an initial population size of 3 300 individuals.

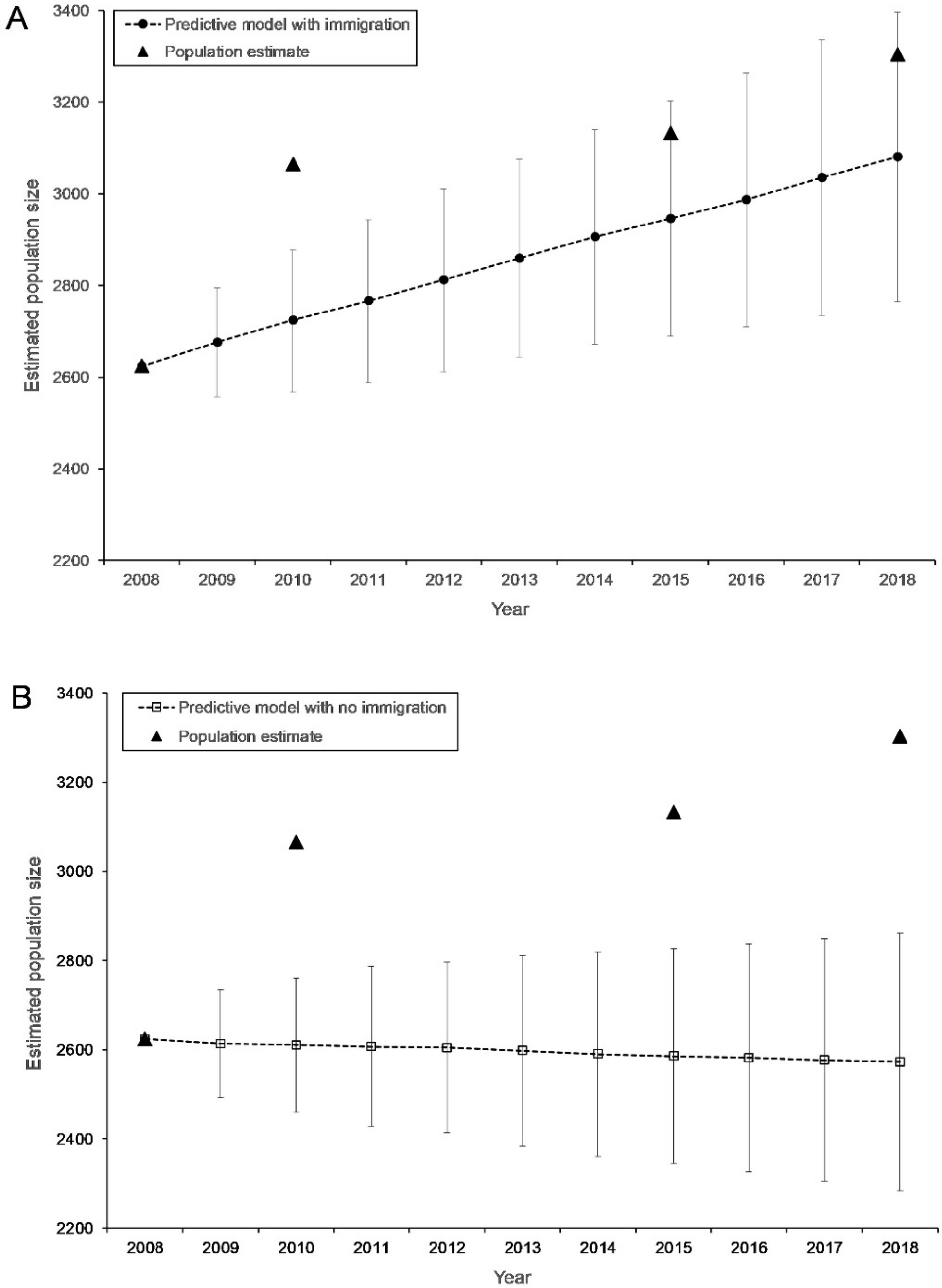

Figure 5.3: From 2008–2018, using Base Model 1 parameters with a starting NZ sea lion population of 2,624 in 2008 (based on a pup count of 583 and pup multiplier of 4.5), the predictive model population trajectories and standard deviations with (A) and without (B) immigration, compared with four population estimates (using a pup multiplier of 4.5) based on Campbell Island pup counts in 2008, 2010, 2015 and 2018 (see Figure 5.1).

5.5 Discussion

In PVA models for the Campbell Island NZ sea lion population, the low growth rate in Base Model 1, even in the absence of bycatch, indicates this population may be demographically challenged, and there was negative population growth when immigration was not included. As for the Otago population and recently established population on Stewart Island (Chilvers 2018; McConkey et al. 2002a; McConkey et al. 2002b), it is feasible that immigration, sourced from the Auckland Islands, has contributed to the growth of the Campbell Island population, although the influences leading to immigration are not clear. A greater understanding of immigration and emigration rates for NZ sea lions would be useful, though difficult to obtain. If the Campbell Island population is sustained by immigration, as appears to be the case, variation in dispersal rates would impact the viability of this population and may also impact the dynamics of the source population.

The annual NZ sea lion captures in the SBW fishery before (average 12/year, 2009–2013) and following SLED implementation (average 3/year, 2014–2017) are well below the PBR estimate of 25, indicating that bycatch levels in this fishery are sustainable. The pre-SLED peak of 24 captures in 2010 is also just below the PBR of 25. Confidence in the sustainability of the PBR estimate is increased due to the strong male bias in bycatch recorded for this fishery (Abraham et al. 2016). Doubling the recovery factor (F_r; a ‘safety’ parameter which accounts for unknown biases in estimates) as a means of accounting for male bias in bycatch resulted in a PBR of 50, which well exceeds all bycatch estimates. The sustainability of this bycatch level, with 98% male bias, was supported by PVA analysis indicating positive population growth with this scenario (Model 7, Table 5.3). Lowering either the productivity rate (R_{max}) or F_r resulted in lower PBR estimates indicating that, if this population began to decrease, or the species and/or population status was in decline, the sustainable bycatch level would be lower (Table 5.2). Conversely, if population growth increases and the species and/or population status improves, a higher bycatch level could be sustained.

The PVA modelling supported the PBR evaluation outcomes and allowed a more refined assessment of bycatch impacts, particularly considering the male bias in bycatch (Abraham et al. 2016; Thompson et al. 2015). The scenario representing current bycatch in the SBW fishery (i.e. two males and one female per year), showed no change in population growth compared to the Base Model 1 with no bycatch. There was slightly reduced, although still positive, population growth with 'high' and 'medium' bycatch levels. However, modelling an annual ‘high’ bycatch of 24 sea lions is probably over-representative as, in 22 years of observation, mean estimated captures only exceeded 15 twice; with 24 in 2010 and 21 in 2013 (Abraham and Berkenbusch 2017) (Figure 5.3). Furthermore, these peaks occurred before SLED implementation in this fishery. The impact on the population would increase if more female sea lions were killed in the fishery. The ‘high’ bycatch model of 12 male and 12 female sea lions still showed positive (although low) population growth (Model 5.1, Table 5.3), with this result supporting the PBR estimate of 25 (with assumed equal sex ratio). However, PVA models indicated that, with annual bycatch of 24 adult females, the population would decline (Model 5.2). An annual bycatch above 34 sea lions (with equal sex ratio) could also shift the Campbell Island population from a positive to negative trajectory (Model 6, Table 5.3). Given the apparent importance of immigration as an influence on the dynamics of this population, any change to immigration rates would affect the ability of the population to sustain bycatch or other anthropogenic impacts.

The PVA scenario simulating on-ground management to reduce pup mortality resulted in a larger increase in population growth (Model 8) compared to scenarios with varying levels of bycatch (Table 5.3). The Campbell Island population has the highest recorded early pup mortality for NZ sea lions (Childerhouse et al. 2015; Maloney et al. 2012), attributed to on-island climate and habitat, with pups falling into wallows being a primary cause of death (Maloney et al. 2009; Roberts and Doonan 2016). The modelling supports current management to improve the NZ sea lion population trajectory which focuses on actions to reduce pup mortality in the first eight weeks of life to less than 40% per annum, as well as maintaining mitigation of bycatch impact from the SBW fishery (DoC and MPI 2017). The installation of ramps allowing pups to escape from holes has reduced early pup mortality (Childerhouse et al. 2015), and bycatch mitigation approaches, including the use of SLEDs, have resulted in low bycatch levels recorded since 2013 (Abraham and Berkenbusch 2017; Deepwater Group 2017a; Deepwater Group 2017b).

The foraging ranges of NZ sea lion sub-adult males from Campbell Island overlap with SBW fishery areas (M-A. Lea, unpublished data), which has been reflected in bycatch data. While further foraging data are required, particularly on female habitat use (to date, only four adult females have been tracked from this population), satellite tracking indicated a low likelihood of NZ sea lion overlap with the Campbell Island hoki (*Macruronus novaezelandiae*) trawl fishery, and a moderate chance of adult female overlap with the Campbell Island ling (*Genypterus blacodes*) longline fishery (M-A. Lea, unpublished data). However, there is no reported NZ sea lion bycatch in either fishery

(Abraham et al. 2016) and pinniped species are not considered at high risk from longline operations (Gray and Kennelly 2018). If fishing practices and target species change, and NZ sea lions interact with operations other than the SBW fishery, updated assessments of bycatch impact would be needed. Changes in effort, gear and practices are driven by market forces and finding suitable approaches to manage the factors influencing bycatch requires good data collection and ongoing collaboration among fishers, fishery managers, biologists and gear technologists (Baker and Robertson 2018).

There has been a recent focus on whether reducing global fishing pressure to halt target species overfishing and maximise the value of fisheries may, in turn, improve the protection of non-target species threatened as bycatch. Burgess et al. (2018) showed that many at-risk species would have projected increased population growth if the relevant ‘overfished’ fishery reduced effort and transitioned to a fisheries profit-maximising rate (i.e. ‘Maximum Economic Yield’, MEY, when the sustainable catch level creates the largest difference between total revenue and total fishing costs). In contrast, they concluded that transitioning to a fisheries profit-maximising rate for the Auckland Islands squid trawl fishery (which they assumed was below MEY and would involve increase in effort) would likely increase NZ sea lion population decline due to higher bycatch mortality, although they did not account for other conservation management measures. While increased effort could result in a proportional increase in sea lion bycatch, the number of deaths and subsequent population impact would be less than their projection as the fishery has implemented effective bycatch mitigation, with corresponding evidence that the Auckland Islands population has stabilised over the last decade (Childerhouse et al. 2018; Hamilton and Baker 2015a; Hamilton and Baker 2016a; Roberts et al. 2018), as well as management limits on bycatch numbers (MPI 2017c). For the Campbell Island SBW fishery, target stock accessibility is restricted to a six- to eight-week period coinciding with spawning aggregations (Deepwater Group 2017b). While fishing effort is unlikely to increase due to the relatively low value product compared to alternative options and difficult operational conditions in the Southern Ocean (Richard Wells, unpublished information), bycatch is effectively mitigated, and any fishing effort increase would be unlikely to substantially impact sea lions. Therefore, management actions for at-risk bycatch species that improve survival, such as effective bycatch mitigation, may have consequential trade-off influences on the MEY and sustainability of relevant fisheries.

Given adequate input data, PVA predictions using generic software packages such as VORTEX are surprisingly accurate (Brook et al. 2000). While it cannot be expected that models would provide prediction certainty when population-specific information is lacking, PVA is a useful tool for guiding conservation research and management by identifying key demographic parameters and impacts that may affect a species' survival (Hamilton and Moller 1995). Due to the limited NZ sea lion population data available for Campbell Island, proxy or default values were used for some parameters in this PBR assessment and PVA modelling. To enable a more robust assessment of impacts, accurate and up-to-date population status and demographic data are needed and should be a resource priority. However, this would take years of research when immediate and imminent actions are required to improve the population trajectory of this endangered species. Based on the best current information, this research highlights the importance of increasing pup survival for the NZ sea lion population at Campbell Island, and supports conservation priorities to reduce pup mortality, while maintaining mitigation to reduce trawl fishery bycatch along with high levels of observer coverage to sustain confidence in annual bycatch estimates.

5.6 Appendices

Appendix 5A: Population size estimate for Campbell Island:

Multiplier values, applied to pup count data to estimate pinniped breeding population sizes, are usually population-specific based on demographic data. The size of most increasing polygynous pinniped populations are estimated by using a pup multiplier between 3.5 and 4.5 (Harwood and Prime 1978). Based on the Auckland Islands population, pup multipliers of 4.4 (Chilvers 2012b), 4.5, 5.5 and 6.5 (Roberts et al. 2014b) have been derived for the NZ sea lion, with multipliers from 4.51 to 5.40 previously applied to the Campbell Island population (Roberts and Doonan 2016). In the absence of population-specific demographic data and applying a precautionary approach, a lower factor of 4.5 was selected and, applied to the estimate of 734 pups in 2018 (NZ Department of Conservation, unpublished data), to derive a population size of 3,300 (rounded to the nearest 10).

Appendix 5B: Potential Biological Removal (PBR)

(i) Calculation of minimum population estimate (*N_{min}*) for Campbell Island:
As recommended by (Wade 1998), N_{min} was calculated using:

$$N_{min} = \frac{N}{\exp(z\sqrt{\ln(1+CV(N)^2)})}$$

where N is the total population estimate; z is a standard normal variate (set at the default 20th percentile, 0.842); and CV(N) is the coefficient of variation of the population estimate. In the absence of data on population size variance for the Campbell Island, and consistent with recommendations in Wade (1998), a default coefficient of variation, CV(N), of 0.2 was used. Therefore, based on a population estimate of 3 300, an N_{min} value of 2 793 was calculated.

(ii) Net productivity rate (R_{max}) selection: Where population-specific data is lacking, the recommended default R_{max} for pinniped populations is 0.12 (Moore and Merrick 2011; Wade 1998). However, while population data for Campbell Island shows an increase since low numbers in the 1980s, the trajectory appears to be levelling. Analysis of theoretical productivity rates fitted to pup count estimates from Campbell Island over a 34-year period indicated R_{max} of between 0.02 and 0.12 (Figure 5B.1). An R_{max} of 0.02 was discounted as, even with data uncertainty, it is unlikely pup counts were >100 in the 1980s, and 0.12

was also discounted as being unrepresentative of the population trajectory. An R_{max} of 0.06 was considered a reasonable fit and selected as the base value in PBR calculations, and estimates using R_{max} of 0.04 and 0.08 were also calculated as an indicator of the sensitivity of PBR levels if the population growth rate decreased or increased (Figure 5B.1).

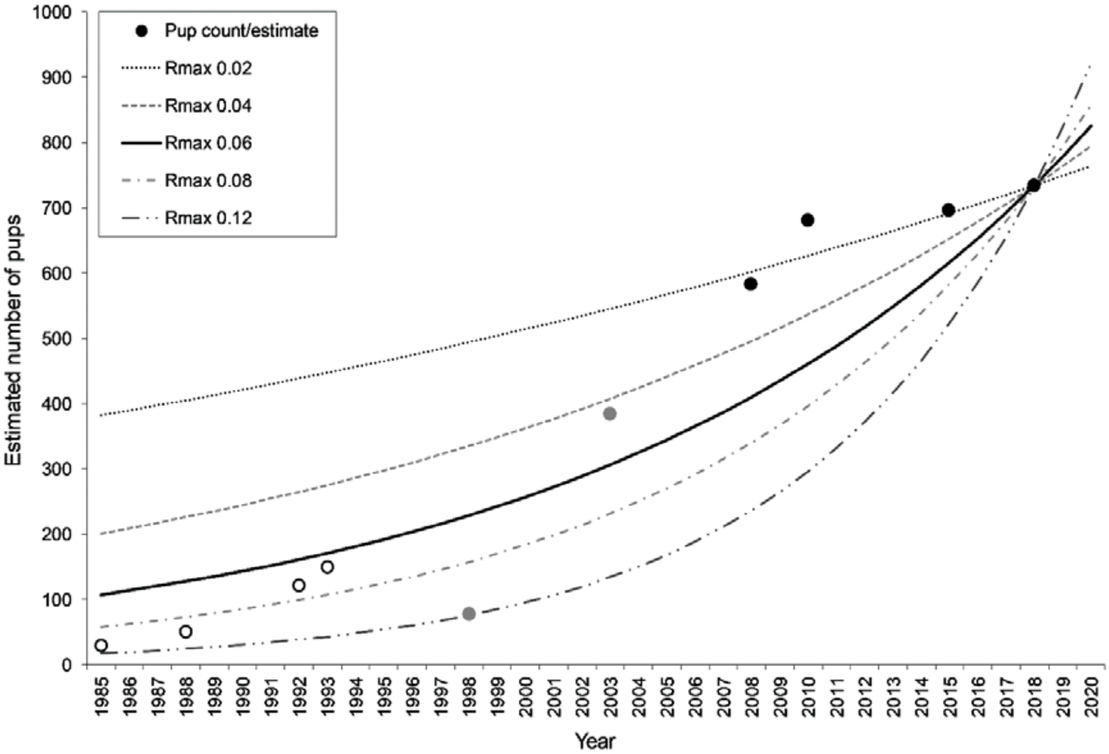

Figure 5B.1: Theoretical productivity rate curves (R_{max} 0.02, 0.04, 0.06, 0.08 and 0.12) for 1985–2020, based on the most recent estimate of 734 pups in 2018, along with NZ sea lion pup count estimates for Campbell Island from 1985–2018. Empty circles indicate direct pup counts, dark grey circles indicate a mixture of mark-recapture and direct counts (noting that 1998 was an incomplete count, McNally et al. (2001)) and, from 2008 onwards, black circles indicate estimates using consistent mark-recapture methodology. Data from Childerhouse et al. (2005) (and references within), Childerhouse et al. (2015) and NZ Department of Conservation (unpublished data).

(iii) Recovery factor (F_r) selection: The application of F_r less than 1.0 allocates a proportion of expected net production towards population growth and compensates for uncertainties that might prevent population recovery, such as biases in the estimation of N_{min} and R_{max} or errors in the determination of population structure (Moore and Merrick 2011). Selecting a lower F_r value also increases the likelihood of obtaining greater long-term population sizes (Lonergan 2011). PBR guidelines recommend a default F_r of 0.1–0.3 for a population listed as endangered or populations known to be declining; 0.4–0.5 for populations that are depleted, threatened or of unknown status; and up to 1.0 for populations at optimal levels, or of unknown status that are known to be increasing (Moore and Merrick 2011; Taylor et al. 2003; Wade 1998). The default F_r of 0.5 for populations of a *threatened* species with unknown (or not declining) population trajectory was considered too high for the *endangered* NZ sea lion, and, conversely, 0.1 was inappropriate as it is recommended for endangered species' populations with a statistically significant ongoing decline. Therefore, a mid-range value of 0.3 was selected for the Campbell Island population which is above the recommended critical abundance threshold of 1 500 and has an increasing trajectory (Taylor et al. 2003). As an indicator of the sensitivity of PBR to changes in F_r, PBR estimates were also calculated for F_r values of 0.2 and 0.4.

Appendix 5C: Population Viability Analysis

(i) Inbreeding depression:

The default values recommended for VORTEX applied the combined mean effect of inbreeding on fecundity and first year survival by simulating 50% of the genetic load due to recessive lethal alleles and a reduction in first-year survival among inbred individuals of 6.29 'lethal equivalents' per diploid individual (Lacy et al. 2017; O'Grady et al. 2006).

(ii) Mortality rates for 'Age 1'— adult:

Apart from mortality in the first year, as for modelling undertaken for the Auckland Islands population (Hamilton and Baker 2016a), the mortality estimates modelled for each age class were based on the inverse of survival estimates published graphically in Chilvers and Mackenzie (2010) (with data subsequently clarified by D. MacKenzie) and modified to account for levels of fishing mortality imbedded in them. As these published survival estimates were based on tag re-sights (for age classes 1, 2, 3 and >4 years) from the Auckland Islands sea lion population from 1998–2005 (Chilvers and Mackenzie 2010), they intrinsically included existing levels of fisheries mortality as well as mortality from epizootics. Therefore, for age classes four years and over (i.e.

the age range predominantly killed in fisheries interactions around the Auckland Islands and the non-pup age range reported to die in the 1998 bacterial epizootic event at the Auckland Islands (Wilkinson et al. 2006)) the relevant age-class mortality estimates reported in Chilvers (2012b) were used which had been adjusted to exclude fishing-related and epizootic mortality ((Chilvers 2012a); L. Chilvers, personal communication). For Age 1–Age 3, mortality estimates were averaged from the Chilvers and Mackenzie (2010) data excluding their estimates for 2004 and 2005, which were based on small sample sizes (Hamilton and Baker 2016a).

(iii) Tag re-sight information used to estimate immigration rates of NZ sea lions:

Previous population modelling of NZ sea lions assumed that dispersal between colonies was negligible as this species, particularly adult females, are considered to be philopatric (i.e. they return to breed at their natal site) based on assessments of marked sea lion re-sights (Chilvers 2012b; Chilvers and Meyer 2017; Chilvers and Wilkinson 2008; Hamilton and Baker 2016a). However, pinniped re-sight data analysis is likely to document natal site fidelity and is less likely to quantify dispersal due to the larger search coverage required (Mathews et al. 2011). While philopatry may be the norm for most individuals, some level of migration to and from populations is also likely, and colonisation of a new area providing unequivocal evidence for dispersal (Mathews et al. 2011). Immigration accounts for much of the early growth of new pinniped colonies (Mathews et al. 2011; Stewart et al. 1994), with juvenile males the main dispersing group for polygynous species (Dobson 1982; Greenwood 1980). For NZ sea lions, there is evidence of dispersal from re-sights of bleach-marked, breeding males (Robertson et al. 2006), satellite-tracking of males (Geschke and Chilvers 2009), re-sights of tagged individuals moving between breeding locations (MPI 2018a), and colonisation of new breeding locations (Chilvers 2018; McConkey et al. 2002a; McConkey et al. 2002b). Furthermore, while it would be expected that species with a high degree of philopatry would have strong genetic differentiation between breeding colonies (Dickerson et al. 2010; Grandi et al. 2018), mitochondrial DNA and nuclear loci analyses have shown a lack of population structure between NZ sea lion breeding colonies, indicating that, from a genetic perspective, they could be treated as one population (Collins et al. 2017). Therefore, the population modelling for Campbell Island included immigration of adult males and females based on the following information.

There has been no permanent human presence on Campbell Island following the de-staffing of the government meteorology station in 1995 (Quayle 1995). NZ sea lion tag re-sight effort has been variable with opportunistic records when researchers working on other species visited the island and more concentrated efforts during research trips focussed on populations surveys of NZ sea lions. I analysed publicly available re-sight data to determine the minimum frequency of movement between breeding islands and to provide an indication of potential immigration rates. Peaks in re-sights have occurred when NZ sea lion pup counts have been undertaken (e.g. 2003, 2008, 2010, 2015) (Figure 5C.1).

The NZ sea lion population at Otago has been monitored since the first pup birth in 1994 and, up to 2001, as many as nine migrant females were recorded at Otago with ages ranging from one-year-old to adult (McConkey et al. 2002b). From 1995–1999, 113 migrant males and four migrant females were identified across a range of age classes (McConkey et al. 2002a) (Table 5C.1).

From tag re-sight data from 1987–2016 (MPI 2018a), of 456 known-sex individuals, 79% (360) were male and 21% (96) were female (Table 5C.2). Of 73 individuals tagged on the Auckland Islands and re-sighted at Campbell Island, 15% (11) were female and 85% (62) were males.

Table 5C.1: NZ sea lion dispersal data from Otago, New Zealand (columns 1–3) used to estimate the number of immigrant individuals per age class for males and females (columns 5 and 6) in the Population Viability Analysis (PVA) models for Campbell Island. Data in column 2 is the age distribution of 40 new identified males (not including pups) sighted from 1995–1999 in Otago from Table 1 in McConkey et al. (2002a). The proportions in column 3, based on the data in column 2, were used to determine the age distributions of males and females in columns 5 and 6.

Of 40 new identified sightings at Otago, 1996–1999:			Modelled immigration of 60 sea lions/year to Campbell Is.:		
Column 1: age	Column 2: # identified	Column 3: proportion	Column 4: age	Column 5: males (M)	Column 6: females (F)
1 year	10	0.25	1–2 years	12	2
2 years	10	0.25	2–3 years	13	2
3 years	3	0.075	3–4 years	4	1
4 years	3	0.075	4–5 years	4	1
5 years	4	0.10	5–6 years	5	2
≥6 years	10	0.25	6–7 yrs (M)/6+ yrs (F)	3	2
Total	**40**	**1.00**	7–8 years	3	–
			8–9 years	3	–
			9+ years	3	–
			Total	**50**	**10**

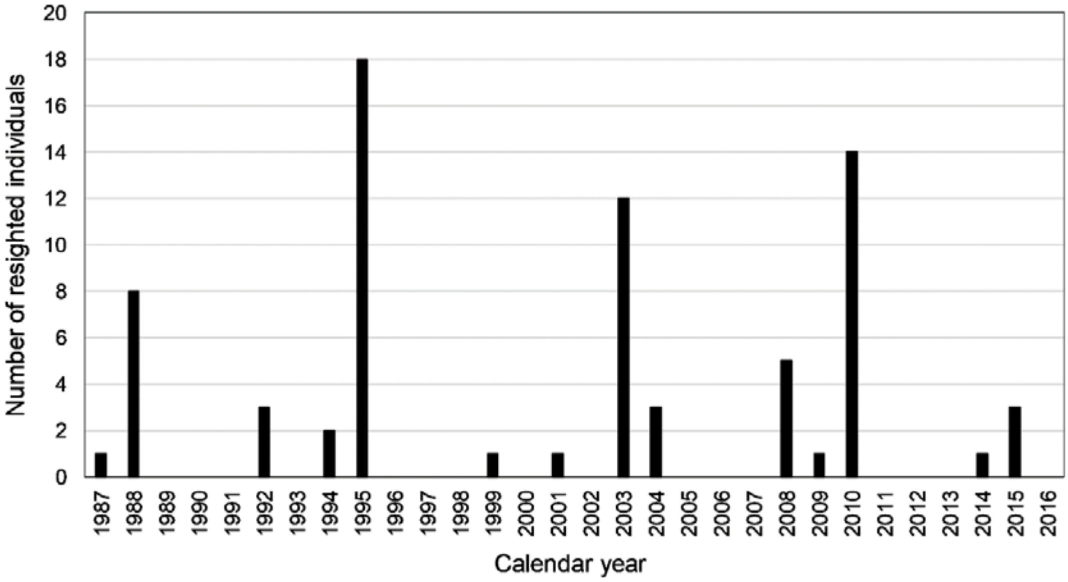

Figure 5C.1: Distribution across years of NZ sea lion re-sights at Campbell Island for individuals that were tagged at other locations. Tag re-sight records for 1987–2016 from MPI (2018a).

Table 5C.2: Records of NZ sea lions tagged at one location and re-sighted at another location. Tag re-sight records for 1987–2016 from MPI (2018a).

Location(s) tagged	*Location(s) re-sighted*	*Male*	*Female*	*Unknown*	*Total*
Auckland Islands	Campbell Island	62	11	0	73
Auckland Islands	The Snares, South I., Stewart I. or Macquarie I.	153	40	21	214
Campbell Island	Auckland Islands	43	15	0	58
Campbell Island	South Island	1	0	0	1
The Snares, South I. or Macquarie I.	Auckland Islands	95	29	0	124
South I., Stewart I. or The Snares	South Island or The Snares	6	1	10	17
Total		**360**	**96**	**31**	**487**

Chapter 6 — General Discussion and Future Directions

Global review of technical mitigation to reduce marine mammal bycatch

Worldwide, incidental bycatch in fishing operations continues to be one of the biggest threats to marine mammal species (Jaiteh et al. 2013; Read 2008; Reeves et al. 2013). Bycatch problems are often best alleviated by implementing a suite of management measures including operational protocols and appropriately designated spatial and/or temporal closures, with the deployment of effective technical mitigation measures (devices or gear modifications) an essential component of many ‘mitigation suites’. In Chapter 2, I provide the first comprehensive global review of technical mitigation measures designed to reduce marine mammal bycatch in commercial trawl, purse seine, longline, gillnet and pot or trap fishing gear. I assessed mitigation testing, effectiveness, and where relevant, operational deployment, and present a synthesis of best practice mitigation and areas requiring greater attention.

When developing and evaluating the effectiveness of a mitigation measure for reducing marine mammal bycatch, the issues and steps to consider include:

- Understanding both the ecology and behaviour of bycatch and target species and how the fishery overlaps with, interacts with and/or targets the bycatch species;
- Obtaining sufficient pre-implementation information, using adequate levels of independent observer coverage, to record bycatch levels and enable an assessment of bycatch reduction;
- Having a clear, quantitative target to measure bycatch effectiveness, e.g. zero bycatch or bycatch below an identified, sustainable threshold (Dawson et al., 2013, Read, 2013);
- Having a rigorous scientific demonstration of effective reduction in bycatch mortality, ideally against a control of no-deterrent (which can often be difficult to implement due to the ethics of allowing bycatch in the control treatment), together with maintenance of target catch quality and quantity;
- Most often, tailoring and refining mitigation to each bycatch problem and undertaking species and fisheries-specific efficacy testing;
- Managing potentially conflicting mitigation outcomes if multiple species are impacted by a fishery and ensuring an integrated approach so that techniques

that reduce bycatch of one species do not exacerbate risks for another (Gilman et al. 2019);

- At-sea testing and monitoring to ensure operability and effectiveness, identify any required adjustments and facilitate continuing efficacy in an operational environment;
- Adherence to regulatory requirements and post-implementation auditing to ensure compliance with mitigation implementation and that the correct specifications are met;
- Assessing the impact of bycatch levels, any reduction in bycatch mortality and the sustainability of bycatch on the affected species population; and,
- Ongoing and regular reviews of mitigation effectiveness over time to ensure efficacy is maintained.

The involvement and engagement of key stakeholders, particularly the fishing industry, scientists and resource managers, is crucial for the most effective, functional mitigation design and to ensure uptake and compliance with mitigation implementation and adherence to best-practice (Hamer et al. 2009; Read 2013). Furthermore, multi-jurisdictional and/or multi-government commitment may be needed so that relevant measures can be implemented across the full ecological range of relevant bycatch species (Read 2013).

A mitigation technique is often deployed and made mandatory in a fishery with limited evidence that it is effective, especially when a suite of measures are implemented at the same time and there is an observed bycatch reduction which, rightly or wrongly, justifies continuation of the employed measures. For some gear types and taxa, there are currently limited technical options that provide strong evidence of an effective reduction in marine mammal bycatch. For example, there are no proven and reliably effective technical solutions to reduce small cetacean mortality in trawl nets, although the use of loud pingers is promising (Northridge et al. 2011). Catch and hook protection devices may reduce marine mammal interactions with longline hooks (Hamer et al. 2015; Moreno et al. 2008; Rabearisoa et al. 2015), although more fisheries-specific trials are needed. Solutions are needed to reduce the bycatch of pinniped and small cetacean species that are not deterred by pingers and continue to be caught in static gillnets. Large whale entanglement, particularly of large baleen species, in pot and trap buoy lines and other static gear requires urgent attention although there is encouraging research on rope-less pot/trap systems (How et al. 2015; Partan and Ball 2016; Salvador et al. 2006). However, acoustic deterrent devices (pingers) have been shown to successfully reduce the bycatch of some small cetacean species in gillnets (Dawson et al. 2013; Kraus et al. 1997; Palka et al. 2008; Reeves et al. 2013), various pot/trap guard designs have reduced marine mammal entrapment (Campbell et al. 2008; Konigson et al. 2015; Noke and Odell 2002) and appropriately designed exclusion devices have reduced pinniped bycatch in some trawl fisheries (Chapter 3; Hamilton and Baker (2015a)).

Effective bycatch mitigation has reduced NZ sea lion mortality

One technical mitigation device that has received substantial focus is the Sea Lion Exclusion Device (SLED) developed, tested and implemented to reduce endangered NZ sea lion mortality in the Auckland Islands squid trawl fishery. The SLED, a solid steel grid before the trawl codend that directs sea lions upwards to a top-opening escape hole, allows target squid species to pass through the grid into the codend, while an additional hood cover and kite reduces target catch loss and ensures the escape hole remains open for sea lions (MPI 2012). Since peaks in sea lion bycatch in the mid to late 1990s, bycatch has been greatly reduced following successful mitigation implementation, including widespread use of standardised SLEDs in this fishery since 2004/05 (Chapter 3). However, despite greatly reduced observed bycatch, with high levels of independent observer coverage, the ability of SLEDs to actually reduce sea lion mortality has been contentious, with assertions by some scientists and conservation lobbyists of high levels of 'cryptic' or unaccounted mortality. This includes concerns that individuals (i) may suffer injuries, particularly brain trauma, from encountering the hard grid that could affect post-escape survival, (ii) may still die but their bodies may passively drop out the escape hole ('carcass non-retention'), or (iii) may successfully exit but drown before reaching the surface ('post-escape drowning') (MPI 2017a).

In Chapter 3, I reviewed a range of trials and research projects undertaken by the NZ government and fishing industry to determine the effectiveness of SLEDs in reducing sea lion bycatch mortality. I concluded that i) the available evidence shows SLEDs are effective in reducing sea lion bycatch, ii) sea lions are able to escape via SLEDs and are unlikely to sustain life-threatening injuries, and iii) SLEDs contribute to reduced rates of observed sea lion mortality in the Auckland Islands squid trawl fishery. In addition, analysis of archived underwater video of Australian fur seals in the Australian midwater blue grenadier trawl fishery using similar Seal Exclusion Devices (SEDs) showed there is no evidence that pinniped carcasses would drop out top-opening escape holes, although interactions (and therefore sample sizes) in this study were low (Hamilton et al. 2019). Any further trials of exclusion devices utilising underwater video, which entails significant investment, needs to ensure that a) while testing regimes should be targeted to address key research questions, methodology should retain general applicability for future, re-purposed assessments, b) records of corresponding metadata should be kept (e.g. fishing depth, weather conditions, trawl speeds, target catch size and quality) and, c) where possible, variables should be standardised to reduce the array of factors potentially confounding data, such as consistency with equipment used, positioning of cameras and lighting.

There remain some uncertainties around quantifying how many NZ sea lions successfully escape trawl nets but are unable to reach the surface before exhausting their breath-holding capabilities, a key component of cryptic mortality. Simulation studies and expert opinion have been used to estimate the probability of post-escape drowning (Middleton 2019) to calculate a 'multiplier' to apply to observed bycatch and better inform bycatch limits set by fisheries management (Meyer 2019). This preliminary assessment estimated the cryptic mortality multiplier to be 1.15 (95% CI, 1.05–1.31) for bottom trawls and 1.60 (95% CI, 1.20–2.63) for midwater trawls (Meyer 2019). While more robust data, particularly for 'carcass non-retention' and 'post-escape drowning', would reduce uncertainties in the cryptic mortality multiplier, obtaining such data would be difficult and expensive.

The most compelling evidence that cryptic mortality is unlikely to be significant is provided by key NZ sea lion demographic parameters, including survival estimates from mark-recapture data, and population viability assessments. An important aspect of assessing the ecological sustainability of fishing operations is through assessment of survival, productivity, immigration and emigration and the relative contribution these make to the population growth rate and trends of a bycatch species. Unsustainable levels of mortality, including from bycatch, would be reflected in negative population growth.

Population Viability Analysis (PVA) of the largest NZ sea lion population on the Auckland Islands (Chapter 4), using the program VORTEX, showed that current bycatch estimates from all relevant trawl fisheries are sustainable following effective bycatch mitigation implementation, although population growth is slow. Modelling indicated that epizootic events causing reduced pup production, especially if more frequent than first reported, may have a greater impact on population growth and provide a valid explanation for the population decline observed from the mid-1990s to late 2000s. This is supported by recent pathology research which concluded *Klebsiella septicaemia* could have historically been, and continues to be, the most important cause of pup mortality and is now considered endemic (i.e. a constant presence) in this population. (Michael et al. 2019; Roe et al. 2015). As current bycatch levels in trawl fisheries operating around the Auckland Islands are unlikely to be having a significant impact on this population, resources should be focussed on alleviating disease and other impacts to improve the population trajectory while still maintaining, and continuing to refine, fisheries bycatch mitigation.

NZ sea lions are also killed as incidental bycatch in the southern whiting trawl fishery operating near their second largest breeding population on Campbell Island, with a strong male bias (98%) in observed bycatch. PVA as well as the Potential Biological Removal (PBR) procedure showed that bycatch levels, particularly following implementation of effective bycatch mitigation measures including SLED deployment, are sustainable for the Campbell Island population (Chapter 5). Models showed that reducing pup mortality through management actions, such as installing ramps in wallows where large numbers of pups drown, would lead to increased population growth. While obtaining more accurate data on population status and demographic parameters for this population should be a priority, this will take many years of research. The PBR and PVA tools demonstrate that contemporary conservation management should continue to focus on increasing pup survival while maintaining mitigation approaches that have reduced bycatch to low levels.

As part of the 5-year Threat Management Plan (TMP) for NZ sea lions (2017–2022) (DoC and MPI 2017), annual monitoring has been undertaken at Campbell Island for the past three breeding seasons (2017/18–2019/20). Monitoring in 2018/19 and 2019/20 reported that pup mortality levels continue to be very high (54% and 81%, respectively), largely attributed to extreme weather events (Foo and Weir 2019; McNutt et al. 2020). If high pup mortality is not adequately alleviated, the ensuing reduced recruitment of young animals into the breeding population will likely result in population decline on Campbell Island in coming years. The TMP identifies the need for a strategy to reduce pup mortality from natural holes. Alleviating the impact of weather events may also require innovative mitigation if pup mortality is to meet the TMP target of < 40% per annum. Furthermore, improved data on disease prevalence is needed to identify the contribution this makes to pup mortality at Campbell Island and to inform conservation management for this population (S. Michael, personal communication).

The conservation and management of NZ sea lions is a highly politicized and often controversial topic in NZ and has received much attention by scientists, media, conservation lobby groups and the general public. Consequently, any article or publication on NZ sea lions, particularly regarding fisheries management, attracts considerable attention and often alternative views are provided from those with competing agendas. Specific examples include responses to published papers from Chapter 3 and Chapter 4 (Hamilton and Baker 2015a - Meyer et al. 2016; Hamilton and Baker 2016a - Robertson 2015). As part of the robust scientific peer process, I provided evidence and arguments to refute and discount the claims made by these authors regarding the conclusions in my original papers (Hamilton and Baker 2015b; Hamilton and Baker 2016b). I also co-authored a rebuttal to Meyer et al. (2017) refuting their conclusion that annual pup production changes were primarily driven by cryptic bycatch of adults in sub-Antarctic trawl fisheries and disputing their claim that SLEDs could obscure rather than alleviate sea lion mortality (Roberts et al. 2018).

While the politicization of conservation issue and community activism can lead to positive and beneficial outcomes if the focus and direction is informed, it can lead to negative outcomes and misdirection of resource priorities if not supported by science and evidence (Haward et al. 2013; Tracey et al. 2013). Recognising the efforts of the fishing industry in seeking to modify fishing gear to mitigate non-target (especially endangered species) mortality in fishing operations is also important. The ongoing willingness of fishers to collaborate with scientists, whose findings may identify conservation issues that could impact on their operations or businesses, is contingent on scientists reporting their findings objectively (Hamer et al. 2008; Hamilton and Baker 2015b).

Reassessment of NZ sea lion conservation status

My thesis, including population modelling of the two largest NZ sea lion populations and evaluation of the current impact of fisheries bycatch following effective mitigation implementation, shows that the conservation status of the NZ sea lion has likely improved over the past decade or so, and a re-assessment under the IUCN classification (IUCN 2012; IUCN 2019) is warranted. The last assessment (Chilvers 2015) strongly focussed on the Auckland Islands population without adequately addressing a full population assessment and did not acknowledge or account for the reduction in fisheries bycatch. In that assessment, the NZ sea lion was uplisted from Vulnerable to Endangered on the IUCN Red List (Chilvers 2015), based on the modelled negative population growth rates published in 2012 (Chilvers 2012b) to predict an ongoing population decline of 72% from 1997/98 to 2029/30.

Encouragingly, NZ sea lion pup production at the Auckland Islands has at least stabilised in recent years, and all other breeding populations are increasing or stable (see Figure 4.1 for Auckland Islands, Figure 5.1 for Campbell Island and Figure 6.1 for Stewart Island and the Otago coast). Furthermore, the species has expanded in distribution across its range, bringing the number of breeding populations to four, with at least six separate breeding locations (Enderby Island, Dundas Island, Figure of Eight Island within the Auckland Islands group; Campbell Island; Stewart Island; Otago coast). While the NZ sea lion population continues to have significant conservation issues, conservation management needs to be planned and implemented within the context of a balanced assessment of the species status utilising fair and reasonable processes.

Whilst the conservation status has likely improved, the rate of NZ sea lion population growth remains suppressed and is lower than rates for populations of other sea lion species exhibiting positive growth[3]. Models indicate slow growth at the two largest populations with rates of approximately 0.01 per annum for both the Auckland Islands and Campbell Island (Chapters 4 and 5). These slow growth rates may be partly attributable to marginal breeding as well as foraging habitat, with evidence that adequate nutrition may be more difficult to obtain for the Auckland Islands population compared to the Otago coast (South Island), partly due to food resources being farther from the breeding colony (Augé 2011; Augé et al. 2011). Juvenile and adult female sea lions from the Auckland Islands spend more time at sea, forage over larger areas and dive deeper and longer than those from Otago (Chilvers 2018; Leung et al. 2013). Reduced nutrition is thought to have been a key driver of the Auckland Islands' population decline (Roberts 2015; Roberts and Doonan 2014), with both adult females and pups exhibiting signs of resource limitation (Roberts and Doonan 2016). There are also indications of diet composition changes for NZ sea lions at the Auckland Islands (Childerhouse et al. 2001; Stewart-Sinclair 2013). While it is unknown whether resource limitations are due to climate and/or fishery-related abundance changes (Roberts 2015), there are signs of changes in ocean climate and oceanography around the NZ sub-Antarctic region (Forcén-Vázquez et al. 2017), with record high ocean water temperatures reported for southern NZ over the 2017/18 summer (BoM and NIWA 2018). It is essential that efforts focus on continuing to understand, and where possible alleviate, key population impacts for NZ sea lions, including disease, climate and any fisheries influences (Roberts et al. 2018).

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[3] Annual growth rate of 0.07 for Californian sea lions (*Zalophus californianus*) found along Californian and Mexican coasts (Aurioles-Gamboa and Hernández-Camacho 2015; Laake et al. 2018); 0.06 for South American sea lions (*Otaria byronia* in northern and central Patagonia, and 0.038 at Falkland Islands 1995-2003 (Cárdenas-Alayza et al. 2016); 0.018 for Steller sea lions (*Eumetopias jubatus jubatus)* in USA (Gelatt and Sweeney 2016).

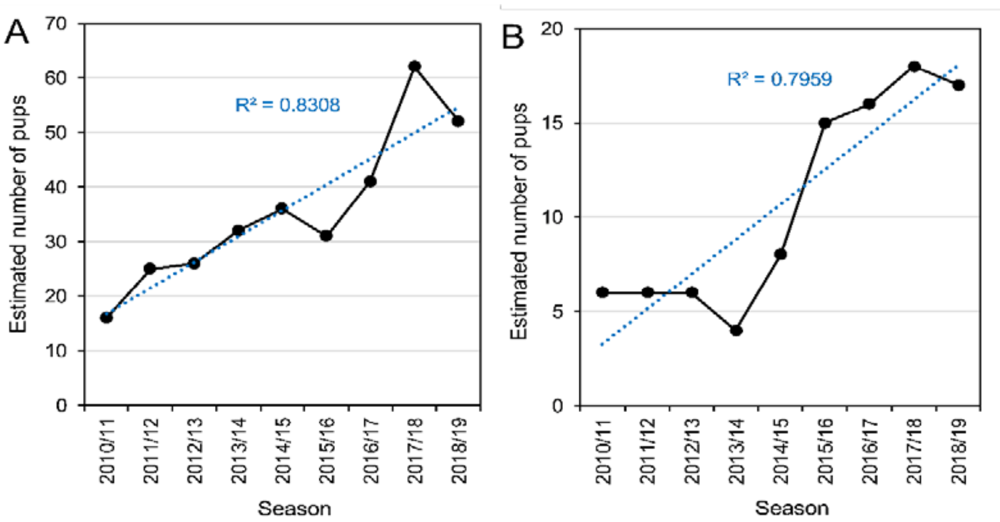

Figure 6.1: Using available estimates from 2010/11–2018/19, the estimated pup production for NZ sea lions at A) Stewart Island (with 2018/19 considered an underestimate) and B) the Otago coast, South Island (note different scales on y-axis). *Data from Roberts and Doonan (2016), MPI (2017b), Department of Conservation (DoC) (2018), Department of Conservation (DoC) (2019), Department of Conservation (DoC) (2020). Fitted lines represent simple linear regressions.*

Future directions

Bycatch of adults is no longer considered the primary impact on NZ sea lion populations. However, disease-induced mortality, particularly the impact of *K. pneumoniae* on pup survival, climate related impacts and breeding habitat 'traps' causing high pup mortality, as well as reduced breeding rates linked to nutritional stress may all be influencing the population trajectory of this species. While addressing potential climate change impacts requires large scale, complex solutions, on-ground actions to ameliorate pup survival will likely improve population trajectories and the robustness of the species conservation status. Therefore, future funding priorities should focus on management options to alleviate pup mortality, such as disease control through treatment (Michael et al. 2019), maintaining ramps in wallows (DoC and MPI 2017) and identifying innovative measures to reduce localised weather impacts.

NZ sea lion bycatch levels in trawl fisheries are likely to remain low now that significant reductions have been achieved through mitigation implementation, particularly the development and implementation of SLEDs. There have been less than 10 observed captures in fisheries per annum since 2012/13 (Abraham and Thompson 2019). However, mitigation still presents challenges to fisheries managers and currently no single measure can reliably eliminate the incidental mortality of sea lions in midwater trawl fisheries.

Seal Exclusion Devices (SEDs), with a similar design to SLEDs, have been used in Australia's blue grenadier trawl fishery to reduce the bycatch of fur seals. However, this large-sized target fish can become clogged on a standard SED grid. An exciting new direction in excluder device development is a modified design with a specialised hinged grid held open at fishing depths, and beyond the diving range of seals, to allow large fish to flow smoothly into the codend. Net binding is needed in conjunction with this device to ensure seals are unable to enter the net during the shot when the device's hinged grid is open. Before hauling, an acoustic release mechanism is used to remotely close the hinged grid to exclude seals from the codend. This modified device is likely to reduce seal bycatch in this fishery, improve the quality of target catch, and could have widespread application in other similar fisheries with large-sized target species. Although preliminary assessment of this specialised device has been positive (see Chapter 3; Hamilton et al. (2019)), further refinement and testing of efficacy is needed. Future trials should have a robust scientific design, preferably with a control, as well as including industry support and engagement to ensure operational effectiveness, uptake and implementation.

Notwithstanding the politicising and controversy associated with fisheries management and the conservation of NZ sea lions, the SLED is a good example of how mitigation development, refinement and testing should proceed, with encouraging bycatch reduction outcomes. Fisheries bycatch has been effectively mitigated, the population decline observed at the Auckland Islands has been halted and is now at least stable, there has been an increase in the species' distribution and there is evidence that their conservation status has improved. Nevertheless, the focus should remain on improving the conservation outcomes for NZ sea lions as their overall population remains lower than historic levels, the population at the Auckland Islands is still lower than what it was in the mid-late 1990s, pup mortality rates are high and population growth rates are low. While the brake should not be applied to monitoring and conservation management of NZ sea lions, it is encouraging that this NZ endemic species can no longer be considered threatened and efforts that have contributed to this should be celebrated. Acknowledging and celebrating conservation gains is important. The return on years of substantial investment in funding, knowledge, dedication and time has resulted in bycatch reduction to sustainable, negligible levels. Acknowledgment of achievements such as these is vital so that policy makers and resource decision-makers can see that investment can achieve useful conservation outcomes and the conservation of imperilled species is worthwhile (Garnett et al. 2018).

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