Population dynamics and anthropogenic threats to New Zealand fur seal (*Arctocephalus forsteri*) in New Zealand

A thesis submitted in total fulfilment of the requirements for the degree of

Doctor of Philosophy (PhD)

in

Veterinary Science

at Massey University, Palmerston North, Aotearoa New Zealand

Alasdair A. Hall, 2025

Abstract

New Zealand fur seals (*Arctocephalus forsteri;* 'NZFS') are New Zealand's most encountered pinniped. However, substantial gaps exist in the knowledge of their abundance and distribution. This study provides NZFS abundance and distribution data for Kaikōura and Banks Peninsula and investigates anthropogenic risks in both locations. Additionally, the thesis undertakes the first nationwide NZFS abundance estimate in ca. 50 years.

The Kaikōura population study was the first since the 2016 earthquake. Kaikōura's NZFS population has grown and spread post-earthquake, with an upper population estimate of 21,560 – 28,327 NZFS in the 2022/23 breeding season. However, pup production at Ōhau Point, the most impacted colony, has not grown, and breeding distribution has changed significantly. Following earthquake damage, State Highway 1 (SH1), which runs close to NZFS colonies, was reconstructed. This study detected an almost fivefold increase in the annual number of NZFS recorded on SH1 from 2012 – 2022, compared to 1996 – 2005. Ten statistically significant NZFS incident clusters were located, representing 89% of the incidents. Cluster location shifted following post-earthquake road reconstruction. Monthly NZFS incident numbers were significantly positively associated with traffic and windspeed, and significantly negatively associated with temperature and rainfall. Road-abutting NZFS breeding explained most of the spatial variation in NZFS incidents.

An abundance estimate of 13,147 – 17,675 NZFS was calculated for Banks Peninsula in 2023/24, and 25 previously unrecorded colonies were assessed. This study considered response strategies for an oil spill impacting Banks Peninsula's NZFS, as the region is classified as 'high risk' for such incidents. Priority response strategies include preventing oil from reaching colonies, and hazing individuals away from waterborne slicks.

From the most recently available count data, a minimum nationwide population estimate of 131,338 – 168,269 NZFS was calculated. Using recent counts and stage-structured population modelling, a more reliable estimate of 181,646 – 239,473 NZFS was calculated, a substantial increase on the most cited nationwide abundance figure, 100,000 NZFS.

This thesis' population findings provide useful baselines and highlight the need for improved NZFS population monitoring. This is particularly important due to the changing human-NZFS relationship, evidenced by the Kaikōura road reconstruction and the risk of oil spills in Banks Peninsula.

Acknowledgements

This thesis would not have been possible without the hard work and support of so many people, to whom I'm very grateful.

Firstly, I would like to acknowledge and thank my supervisors, Louise Chilvers and Jody Weir. Louise - thank you for helping me through every stage of this work, from defining my research objectives, selecting methodologies, showing me how to catch pups, helping me interpret my results, and answering countless questions. Your knowledge, as well as your encouragement for me to challenge myself, not only gave me the confidence to do this, but also ensured I got a lot more out of it than I otherwise would have. Thank you also for everything you did to make the Banks Peninsula fieldwork happen, and for the amazing experience working with sea lions down on Rakiura. To Jody, thank you for providing the initial idea that this research developed from, and for the huge amount of support, knowledge and time you have put into helping me see it through – in particular, all those hours walking up and down Kaikōura's coastline counting pups! Your passion for Kaikōura's marine species is infectious and is a big reason why I embarked on this journey. I'm very grateful for our chance encounter in Kaikōura seven years ago!

I would also like to thank Laura Boren for her generosity with her time, expertise and fur seal knowledge. Laura – much of this thesis builds on your work with fur seals in Kaikōura and Banks Peninsula, and I'm very lucky to have had your guidance, both in the field and out of it, in helping me document the latest chapter of their story.

A huge thank to all the volunteers who helped me at various stages of the work – Devon, Kasey, Emma, Mal, Paul, Nikki, Štefan, Nora, Brad, Nic, Lauren, Guy, Hailey, Grace, Matt, Liz, Diana, Mie, Ally, Anne, Fi, Alex, Katia, Amy, Aaron, Stephen, Mikaylie, Thomas, Wendy, Anna, George, Will, Lucy, Sarah, Julie, Delia, Sara and Benoit. This work would not have been possible without you, and I am very grateful. A particular shoutout to the long-term fieldwork volunteers: Lauren, Brad, Guy and Hailey. Lauren – for all the organisational and admin support both before and during the two field seasons, and for keeping me sane. Brad 'the seal whisperer' Brown – for all your driving and BBQ expertise behind the scenes, Guy – for always being down for a bit of extra for experts, and Hailey – for helping me control Guy. A massive thank you also to Ashley Vidulich and Dr. Jonathan Godfrey for all your hard work on some of the more complex statistical analyses.

Thank you also to Ben and Katie at the Albatross Hostel in Kaikōura for providing me (and the fieldwork volunteers) with a home-away-from-home over many months in Kaikōura. The warm, welcoming atmosphere you provided was something I always looked forward to when returning to Kaikōura, and without your generosity, I'm not sure how I would have made this work. Similarly, thank you to Alison Evans from Christchurch City Council for providing a fieldwork base in Te Oka Bay, and the Chilvers family for the same in Diamond Harbour.

I also wish to acknowledge mana whenua in both Kaikōura and Banks Peninsula for allowing me to conduct this work and access colonies on their land. I would also like to thank the Department of Conservation, and the numerous DOC staff who have been involved in, and facilitated, this work in many ways, in particular Tom MacTavish and Kaitlyn Leeds from DOC Akaroa for their help in planning the Banks Peninsula fieldwork. Similarly, I want to thank Emma Parr of Environment Canterbury (ECAN), and Averil and Kevin Parthonnaud from Pōhatu Conservation Trust for all their help and support planning the fieldwork in Banks Peninsula, and the many landowners in the area who permitted access to fur seal colonies across their land. I also wish to thank Waka Kotahi/the New Zealand Transport Authority (NZTA), and particularly, Jude Ward, for all the help with access to New Zealand fur seal incident data, and the software I needed to complete these analyses. Huge thanks also goes to Chris Lalas for providing me access to his Otago/Catlins NZFS data, and for all of his advice.

I was fortunate enough to receive funding from several bodies that has enabled me to carry out this research, and so I would like to thank The Coastal Restoration Trust, The New Zealand Coastal Society, The Wildbase Research Trust, the Joan Berry Fellowship and Massey University for electing to support this work through their generous grants. I would also like to thank the Massey University Animal Ethics Committee for their consideration and approval of my fieldwork proposal.

I would like to thank my family - my parents, Richard and Ailsa, and my brother, Guy. Mum and dad, there is no doubt that those countless, amazing experiences with wildlife that you gave Guy and I when we were young are the reason that I'm doing this. Mum, your advice, support and love has been, as ever, unlimited, and this research would not have been possible without it. You also made a mean research assistant, driving up and down the Kaikōura coast as I hopped out to collect data, and your proof-reading was, as always, invaluable. Dad, I miss you, but I know you were looking down and laughing as yet another seal pup gave me the slip during the fieldwork. If I become half the researcher that you were, I'll have done okay. Guy, you and mum both encouraged me to take the plunge and go back to studying when I wasn't sure if it was the right move, advice for which I'm very grateful. I also hugely appreciate the not insignificant journeys you made from Vancouver to help me catch pups in both Kaikōura and Banks Peninsula – one of the best seal barbers in the business. Finally, to my partner, Lauren. You've been there through the inevitable ups and downs that come with a PhD - thank you for helping me to keep my chin up, encouraging me to push myself, masterminding so much of the fieldwork organisation, for being the most enthusiastic fieldwork volunteer, and for enduring endless amounts of 'seal chat'. I love you.

Table of Contents

Table of Figures

Table of Tables

Chapter 1 General Introduction

Male New Zealand fur seal on the Kaikōura Peninsula.

1.1 Introduction

Pinnipeds (otariids, eared seals, fur seals and sea lions; phocids, true seals; and the walrus odobenids, Order Carnivora) are amphibious mammals, foraging at sea but coming ashore to complete important processes such as resting, breeding and moulting (Reichmuth et al. 2013). Due to a life history strategy that relies heavily on two different media, pinnipeds are exposed to a complex range of natural (Soto et al. 2004; Gooday and Goldstien 2018; Conway et al. 2022) and anthropogenic (Ling 1990; Boren et al. 2006a, 2008; Bowen and Lidgard 2013) drivers that can exert significant influence on population dynamics and distribution. Historically, pinnipeds around the globe were subjected to unsustainable anthropogenic harvests (Kenyon and Wilke 1953; Hindell and Burton 1988; Ling 1990; Gerber and Hilborn 2001; Hucke-Gaete et al. 2004), and while some species such as grey seals (*Halichoerus grypus*) (Suuronen et al. 2023) and northern elephant seals (*Mirounga angustirostris*) (Rick et al. 2011) have recovered since harvesting ceased others, such as the New Zealand sea lion (*Phocarctos hookeri*) continue to decline (Chilvers 2019) or, like the Hawaiian monk seal (*Neomonachus schauinslandi*), remain at critically low abundances (Robinson et al. 2021).

Understanding patterns of abundance (Laidre et al. 2015; Campera et al. 2020) and distribution (Austin 2007; Robinson et al. 2011) is essential to species conservation and management. However, in many pinnipeds, crucial population level characteristics remain poorly understood (Kovacs et al. 2012; Hupman et al. 2020; Chilvers 2021; Chilvers and Dobbins 2021). Catastrophic events, whether anthropogenic or natural in origin, can highlight missing knowledge to those seeking to understand the event's effects on wildlife in its aftermath (Filkov et al. 2020; Schwing et al. 2020). Equally, the ability to devise effective response plans for such events will be substantially hampered when population parameters are missing (Battershill et al. 2016; Chilvers and Battley 2019).

This thesis aims to provide important information on the population size and distribution of the New Zealand fur seal (*Arctocephalus forsteri*; 'NZFS') in ecologically important areas of the Canterbury region of New Zealand's South Island, and to provide an updated population abundance figure for the species in New Zealand waters. It also explores two anthropogenic threats to NZFS in New Zealand, with an emphasis on how deleterious interactions between NZFS and human infrastructure can be minimised. This introductory chapter presents overviews of NZFS biology and ecology, previous relevant studies of NZFS and some of the anthropogenic risks to the species.

1.2 New Zealand fur seal/kekeno (*Arctocephalus forsteri*)

1.2.1 Taxonomy, biology and contemporary distribution

New Zealand fur seals are one of nine species within the *Arctocephalus* (fur seal) genus (Chilvers 2018), in the family Otariidae, and are the most encountered pinniped in New Zealand (Cowling et al. 2015). The te reo Māori name for NZFS is kekeno, while in Australia they may be referred to as long-nosed fur seal (Shaughnessy et al. 2015; Foo et al. 2019).

NZFS demonstrate strong sexual dimorphism, with males reaching up to two metres (m) long and weighing between 100 – 160 kilograms (kg), while females grow up to 1.5 m and weigh between 30 – 60 kg (Chilvers 2018). As with many pinniped species, NZFS are polygynous colonial breeders (Carey 1991; Caudron et al. 2009), with bulls (breeding males) controlling a harem of cows (breeding females) throughout the breeding season, and vociferously defending their territories (Miller 1974). Males achieve sexual maturity between 5 – 8 years of age, although it is uncommon for them to control breeding territories until between the ages of 8 – 15 (Mckenzie et al. 2007). Females sexually mature between the ages of 4 and 6, and the oldest known female breeding age is 22 (Chilvers 2018). NZFS have no natural terrestrial predators in New Zealand, but they may be hunted at sea by male New Zealand sea lions, some shark species, orca (*Orcinus orca*) and leopard seals (*Hydrurga leptonyx*) (Lalas and Harcourt 1995).

NZFS are non-migratory (Chilvers 2018), and females show distinct philopatry not only to breeding colonies (Stirling 1971), but to specific areas within a colony (Chilvers 2018). Currently, the range of NZFS includes mainland New Zealand, as well as its offshore and sub-Antarctic islands, South and Western Australia, Victoria and Tasmania (Crawley and Wilson 1976; Dix 1993; Lea and Hindell 1997; Bouma et al. 2008; Dussex et al. 2016). Within mainland New Zealand, there are substantially more NZFS on South Island compared to North Island. However, prior to human arrival in New Zealand, NZFS likely bred around much of the coastlines of both main islands, as well as the country's offshore and sub-Antarctic Islands (Lalas and Bradshaw 2001).

1.2.2 Anthropogenic exploitation and recovery

Many pinniped species suffered from extreme anthropogenic exploitation in the form of fur and blubber harvesting, with several, such as the northern elephant seal, driven to near extinction (Stoffel et al. 2018). While bans on commercial sealing in many parts of the world have allowed some species' numbers to recover (Rick et al. 2011; Stoffel et al. 2018; Suuronen et al. 2023), others remain at low abundances (Robinson et al. 2021) or continue to experience population declines (Chilvers 2019).

NZFS were a food source for early Polynesian settlers, resulting in their extirpation from New Zealand's North Island by 1500 AD (Dussex et al. 2016), and mainland breeding was restricted to the western and southern coasts of South Island when Europeans arrived (Dussex et al. 2016).

In the 18th and 19th centuries (Hoelzel 1999; Schultz et al. 2010; Weinberger et al. 2021), NZFS were subject to uncontrolled anthropogenic harvesting which drastically reduced their population size (Emami-Khoyi et al. 2018). There could have been as many as three million NZFS in New Zealand prior to human arrival, with the current population representing roughly 6% of this figure (Emami-Khoyi et al. 2018). This intensive and unregulated harvest began in 1792 and was mostly over by 1830 due to resource exhaustion (Lalas and Bradshaw 2001), however, sporadic harvesting continued, with a final open season granted in 1946 (Sorensen 1969). From 1792 – 1946, Ling (1990) estimates 329,620 fur seal skins were taken from the New Zealand mainland, the Chatham Islands, Snares Island, Stewart Island, and islands in the Foveaux Strait. The combination of subsistence hunting by Polynesian settlers followed by commercial European harvesting is thought to have reduced NZFS numbers to less than 1% of their pre-human population size (Emami-Khoyi et al. 2018).

In New Zealand, NZFS are now protected under the Marine Mammal Protection Act 1978, and, since commercial hunting ceased, their numbers have increased, and they have recolonised areas within their former range (Wilson 1981; Dussex et al. 2016). Isolated refugia in south-western South Island were likely places from which recolonisation began (Dussex et al. 2016), with NZFS spreading northwards (Street 1964; Dix 1993; Taylor et al. 1995; Boren 2005; Bouma et al. 2008). An observed pattern in this recolonisation is the use of an area first as a haul-out for adult seals, and later as a breeding site (Dix 1993; Ryan et al. 1997; Lalas and Murphy 1998).

Roux (1987) devised a four-stage framework for *Arctocephalus* species recolonization, which has effectively described this process in several fur seal species (Carey 1998; Lee et al. 2018). The first stage, ‘survival’, spans the period from the end of exploitation to breeding, when surviving individuals ensure the population’s endurance. During this phase, the intrinsic rate of increase *(r)* is < 5%. (Roux 1987). Next comes ‘establishment’, a phase during which breeding is confined to founder colonies (Roux 1987). At this stage, *r* is between 5 – 10%. The third stage, 'recolonization', is where numbers grow rapidly, driving the foundation of new colonies as space is exhausted in founder colonies, with $r \leq 15\%$. Finally, during the 'maturity' phase, the rate of increase declines due to density-dependent factors (Roux 1987).

NZFS typically establish new colonies close to existing ones (Mattlin 1978a; Roux 1987; Bradshaw et al. 2000b), suggestive of a spill-over effect (Bradshaw et al. 2000b). This likely represents a compromise between NZFS' philopatry (Stirling 1971) and density-dependent factors that may negatively influence breeding success as colonies grow (Bradshaw et al. 2000b). This phenomenon has also been observed in other fur seal species (Milano et al. 2020a).

The Snares Islands, ~200 km south of South Island, were among the first to experience rapid growth NZFS recolonisation in the 1950s and 1960s, before achieving population stability, perhaps due to spatial limitations (Carey 1998). After northwards colonisation through Stewart Island and south-western South Island (Wilson 1981; Lalas and Bradshaw 2001), Open Bay Islands, off the west coast of South Island, was one of the first larger colonies near the mainland to re-establish, with numbers stabilising in the mid-1970s (Lalas and Bradshaw 2001). In the late 1990s and early 2000s, studies found annual colony growth rates at sites around the South Island to be around 20-30% (Lalas and Harcourt 1995; Taylor et al. 1995; Bradshaw et al. 2000b; Lalas and Bradshaw 2001). Between 2002 – 2005, Boren et al. (2006b) found an exponential growth rate of 32% per annum at Ōhau Point (Kaikōura) and 47% at Te Oka Bay (Banks Peninsula). As well as continued expansion northwards (Dix 1993; Bouma et al. 2008), Watson et al. (2015) also found evidence of ongoing growth at established colonies around Stewart Island.

Longitudinal studies show that three NZFS colonies on the west coast of New Zealand's South Island (Cape Foulwind, Wekakura Point, and Taumaka Island) have recently experienced declines in pup production (Roberts and Neale 2016). For example, at Wekakura Point, pup numbers declined by 80% from 1996 to 2016. In these colonies, low pup production years coincided with years where pup masses were also low, suggesting problems with nutrition, however, the causes of the long-term decline are unknown (Roberts and Neale 2016).

1.2.3 Population abundance estimates

There have been difficulties obtaining accurate abundance and population data for some pinniped species (Kovacs et al. 2012), in part due to inconsistencies in monitoring programs (McIntosh et al. 2018a). NZFS is one such species, and, as a result, there is no current reliable estimate for the abundance of NZFS in New Zealand waters.

Pup production counts are the most reliable ways to estimate pinniped population abundances, as pups are the only instantly recognisable age cohort, and because they are largely restricted to land until they wean (Berkson and DeMaster 1985). Thus, in theory, the pup cohort can be counted in its entirety, whereas substantial numbers of post-weaners could be at sea when assessments take place (Berkson and DeMaster 1985). Different approaches to NZFS pup counts include direct counts from the ground (Watson et al. 2009), imagery from unmanned aerial vehicles (UAVs/drones) (Gooday et al. 2018), photography from planes (Taylor 1982; Baker et al. 2009), and counts from boats (Taylor et al. 1995). Direct counts can give an idea of relative population sizes and temporal trends, however, they may not provide an accurate pup population size estimate, particularly at larger sites (Watson et al. 2009). This is because they do not involve thorough sweeps through a colony to flush out individuals hidden from the counter's vantage point, and because pups may die before counts occur (Shaughnessy et al. 1994; Lalas and Bradshaw 2001).

Mark-recapture currently provides the most accurate method for estimating NZFS pup populations (Mattlin 1978b; Boren et al. 2006b; Watson et al. 2009; Chilvers 2021). This method typically involves marking a pre-determined subset of pups with a haircut (Boren et al. 2006b), flipper tag (Roberts and Neale 2016), or stock marker (Chilvers 2021), and then comparing the ratio of marked to unmarked pups during a colony walk-through, after sufficient time has passed to allow mingling of marked and unmarked pups (Boren et al. 2006b; Gooday 2016; Chilvers 2021). From this ratio, a Peterson estimator can be used to estimate pup production (Boren et al. 2006b; Gooday 2016; Chilvers 2021). Mark-recapture has drawbacks, including being resource and time intensive (Gales et al. 2000) and requiring researchers to be able to access colonies (Chilvers 2021). The technique can also be stressful to seals, and human presence can cause displacement and stampedes, which may result in mother-pup separation and/or pup injury and death (Mattlin 1978b). As such, working with NZFS, Watson et al. (2009) developed calibration indices so that direct counts can be used to estimate absolute pup numbers. They used comparisons between pup numbers recorded in mark-recapture protocols and direct counts at 10 colonies to create two conversion ratios, each applicable to different terrain types. These indices, together with their error ranges, provide more accurate estimates of absolute numbers than unconverted direct counts, although they remain less accurate than mark-recapture (Watson et al. 2009). Additionally, the indices calculated by Watson et al. (2009) are habitat specific, and the authors caution against applying them to direct counts below those recorded in their study. One of the assumptions of mark-recapture is that the population is closed to death, birth and migration (Seber 1982), which means pups that die before the mark-recapture survey takes place need to be independently evaluated and added to the number of live pups recorded to obtain an overall estimate of production (Watson et al. 2009).

In Australia, in the 2013/14 breeding season, 20,426 NZFS pups were recorded in South Australia (Shaughnessy et al. 2014), and 3,518 pups were recorded in West Australia in 2011/12 with smaller populations in Tasmania and Victoria. Using an established multiplier to convert pup production estimates into population estimates (Goldsworthy and Page 2007), Chilvers and Goldsworthy (2015) reported there to be ~117,400 NZFS in Australia.

Other than long-term data sets for three colonies on the West Coast of South Island, monitored regularly since 1990/1991 (Roberts and Neale 2016) and a similar longitudinal study in the Otago region, on the south-east coast of South Island (MacDiarmid and Lalas 2014), NZFS population monitoring in New Zealand has largely involved sporadic local or regional scale counts (Boren et al. 2006b; Bouma et al. 2008; Chilvers 2021). For example, Chilvers (2021) recently provided an estimate of 14,000 – 24,000 NZFS in lower Fiordland. This estimate was considerably higher than previously thought (Baird 2011), and suggests estimates from other regions which have not been reassessed in some time may no longer be reliable.

For over 20 years, total population estimates for NZFS across Australia and New Zealand have been listed as 200,000 (Harcourt 2001; Chilvers and Goldsworthy 2015). However, the recent population estimate for Australia (117,400; Chilvers and Goldsworthy 2015), combined with previously recorded high growth rates at New Zealand colonies (Boren et al. 2006b; MacDiarmid and Lalas 2014), suggests this range wide abundance estimate is likely unreliable. An updated population estimate for NZFS in New Zealand is required, however, some locations, such as Banks Peninsula, have not been assessed in approximately 20 years

(Allum and Maddigan 2012), while other populations such as on those on the Antipodes Island (New Zealand sub-Antarctics), have not been assessed in 40 years (Taylor 1992).

Due to the practical difficulties associated with assessing species abundance across large geographic areas, modelling approaches are commonly adopted (Gales and Fletcher 1999; Acevedo et al. 2014; Taylor et al. 2022). In addition to estimating current species abundances, it is also possible to project both future (Lyngdoh et al. 2018; Bino et al. 2020) and past (Emami-Khoyi et al. 2018) population sizes. Age-structured models form the basis of nearly all population modelling involving vertebrate species that have a birth pulse (where all or most births occur at a particular time of year) (Schaub and Kéry 2022). Stage-structured models (Lefkovitch 1965) are a derivation of age-structured models (Leslie 1945), and both are based on observations of changes in key vital rates among individuals as they progress through life. For example, it is common for mammal species to experience relatively high juvenile mortality rates, which decrease during prime years and then increase as individuals age (senescence) (Beauplet et al. 2006; Knape et al. 2011; Condit et al. 2014). Similarly, female reproductive output often varies with age, increasing as individuals hit sexual maturity, before decreasing again with senescence (Beauplet et al. 2006; Descamps et al. 2008; Knape et al. 2011). To use age or stage-structured models it is necessary to have data for key vital rate parameters – fecundity and survival – at different ages/stages. Lefkovitch (1965) noted that such parameters often remain relatively constant once organisms reach adulthood, meaning that age classes could be collapsed into stages – for example, egg, larva, pupa and adult (Zheng et al. 2019). As well as simplifying the matrices required for calculating population size estimates, stage-structured models are useful when vital rate data for the relevant species are lacking for every age group but can be estimated for life phases (Gales and Fletcher 1999). Both age and stage-structured models can enable comprehensive quantitative assessments of a population by allowing examinations of, for example, the contributions of certain cohorts to population growth (Chilvers 2012; Condit et al. 2014) and enabling forecasting of population persistence into the future (Wittmer et al. 2010).

When projecting future population abundances or rates of population change, models can consider density-dependence (Brook and Bradshaw 2006). Density-dependence refers to the inability for animal populations to grow limitlessly over very long periods of time, due to resource limitations (Guthery and Shaw 2013) which take effect as the population expands. This is considered within Roux' (1987) fur seal colonisation framework where, during the maturity phase, rates of population growth slow due to density-dependent factors. In fur seals, such density-dependent factors include increased mother-pup separation, interactions with aggressive animals, competition for food (Bradshaw et al. 2000b) and disease transmission in crowded colonies (Berón-Vera et al. 2004). In several locations within their range (Carey 1998; Campbell et al. 2014), it appears that NZFS may have reached their carrying capacity due to a lack of suitable space for breeding sites to expand into, while MacDiarmid and Lalas (2014) believe that NZFS population growth in Otago may be limited by prey abundance.

Demographic and environmental stochasticity are other elements that can be included in population models to improve their reliability (Schaub and Kéry 2022). Demographic stochasticity is caused by random events of individual reproduction and mortality, leading to fluctuations in population growth rate (Knape et al. 2023). By contrast, environmental stochasticity refers to temporal fluctuations in reproductive and mortality rates of all members of a population, causing the population growth rate to fluctuate (Alamaraz et al. 2022). While the effects of demographic stochasticity are generally associated with small populations, environmental stochasticity can impact populations of all sizes (Lande et al. 2003), and both can have substantial impacts on models of population survival.

1.2.4 Approaches to population modelling in pinnipeds

Age and stage-structured models have been used to answer population level questions for many pinniped species (Punt and Butterworth 1995; Trites and Larkin 1996; Gales and Fletcher 1999; Chilvers 2012; Lowry et al. 2014; Meyer et al. 2015; Wege et al. 2016; Thomas et al. 2019). Given that pup production estimates are the most reliable way to estimate pinniped abundances (Berkson and DeMaster 1985), age or stage-structured models are often combined with lifetables to create multipliers for converting pup production estimates into total population estimates (Hauksson 2007; Russell et al. 2019; Chilvers 2021; Hammond et al. 2021). For example, a conversion factor derived from a life table created for NZFS in South Australia (Goldsworthy and Page 2007) has been used to convert pup counts into population estimates for this species in various locations across its range (Boren 2005; Campbell et al. 2014; Chilvers 2021). Such multipliers can only provide relatively coarse population estimates, with wide confidence intervals (Chilvers 2021). Alternatively, population models can be fitted to count data (Thomas et al. 2019), allowing modelling of density-dependent effects, and the inclusion of datasets other than pup counts (for example, adult seals) (Hammond et al. 2021). Given that population and demographic data are lacking for many pinniped species (Kovacs et al. 2012; Hamilton et al. 2019; Sepúlveda et al. 2020), it is common for such information to be utilised from other areas within the species' range (Boren 2005; Campbell et al. 2014; Chilvers 2021) or from similar species (Gales and Fletcher 1999). Even in comparatively well studied species, such as northern elephant seals, it has been necessary to use demographic parameters from southern elephant seals (*Mirounga leonina*) (Lowry et al. 2014) to generate the number of life tables required for modelling.

1.2.5 Colony dynamics and breeding behaviour

Many, but not all, pinnipeds are colonial breeders (Anderson and Harwood 1985). While there are some species that are predominantly solitary breeders, for example, leopard seals (Rogers 2009) others, particularly sea lions, only adopt colonial breeding above certain population size thresholds (Cassini and Fernández-Juricic 2003; Maloney et al. 2012).

A high degree of colonial philopatry is a common characteristic among otariids (Lunn and Boyd 1991; Chilvers and Wilkinson 2008; Kiyota 2021), including NZFS (Stirling 1971). For NZFS, breeding season preparations begin between September and November when bulls come ashore to claim territories, with females arriving a few weeks later to give birth (Stirling 1971; Miller 1975; Boren 2005). Post-partum, females remain close to birth sites for roughly 10 days (Harcourt 2001), and oestrous females usually mate around one week after giving birth (Goldsworthy and Shaughnessy 1994; Harcourt 2001). After the breeding season, the number of adult males at the colony drops (Miller 1975; Boren 2005), with both adult and subadult males often hauling out away from summer breeding sites (Crawley and Wilson 1976).

Phocids and otariids demonstrate different lactation strategies (Schulz and Bowen 2004). Due to their generally larger size and associated fat reserves, phocid mothers typically remain ashore for their entire, shorter, intensive lactation period, fasting as their pup weans (Costa and Trillmich 1988), while otariids alternate between periods ashore suckling their pups and days spent foraging at sea, over a longer period (Costa and Trillmich 1988; Harcourt et al. 2002; Boren 2005). The most cited median lactation duration for female NZFS is 294 (± 6) days, with weaning typically occurring between September and October in the year following the breeding season in which pups were born (Stirling 1971; Harcourt 2001). This timeframe is typical of otariids, with lactation durations for species in this family generally lasting between 10 – 12 months (Wege et al. 2014). Boren (2005) noted a longer mean lactation duration of 340 (± 9) days for NZFS at Ōhau Point in Kaikōura, which could be associated with the proximity of reliable food sources in the Kaikōura Canyon. Boren (2005) suggests that, with plentiful prey resources nearby, female NZFS may be able to stay in better body condition while lactating, allowing them to extend the weaning period without severe energy penalties, and thus wean pups that are in better body condition themselves (Boren 2005).

Aged 3 – 4 months, NZFS pups lose their black juvenile pelage, and gain their waterproof adult pelage (Chilvers and Goldsworthy 2015). Of the pups born in a breeding season, approximately 20% will die before reaching 50 days old (Mattlin 1978b; Lalas and Harcourt 1995) and approximately 40% will die before 300 days, with starvation responsible for up to 70% of pup mortality to 50 days (Mattlin 1978b). Other common natural causes of death for fur seal pups include infection (Duignan 2000), colony density and predation (Harcourt 1992). Boren (2005) found that pup mortality at Ōhau Point in Kaikōura was low to 50 days at 3% and was mostly attributable to emaciation. Post 50 days of age, 66.7% of pup deaths were caused by vehicle collisions (Boren 2005). Assessing mortality rates and causes of death is limited by the fact that many pup carcasses are likely lost to the sea, scavenged or fall into rock crevices, and thus never counted nor examined

1.2.6 Global threats to pinnipeds

Perhaps the most overt threat to pinnipeds is hunting. As well as illegal killings (Li et al. 2010) legal pinniped hunts still occur in several countries (Kirkman and Lavigne 2010; Nunny et al. 2018; Levy 2020). Such hunts are performed for reasons ranging from subsistence (Nelson et al. 2019; Levy 2020), to harvesting pelts for sale and export (Levy 2020) to the perception of fisheries protection (Nunny et al. 2018). The scales of such operations differ vastly by location and purpose (Levy 2020). For example, the management plan for the annual commercial seal hunt in Canada targeting harp seals (*Pagophilus groenlandicus*), permitted a total harvest of 975,000 seals over three years from 2003 – 2005, with a maximum of 350,000 in any given year (Stenson and Upward 2020), and has been criticised as being unsustainable (Levy 2020). By contrast, the subsistence hunting of endangered species such as ringed (*Pusa hispida*) and bearded seals (*Erignathus barbatus*) by Inuit in Alaska is thought to be conducted at a sustainable scale (Nelson et al. 2019).

Even when not directly targeted, pinnipeds are at risk from other anthropogenic harvests through bycatch (Reeves et al. 2013). Between 1990 – 2011, an estimated 66% of global pinniped species had been recorded in gillnet bycatch alone (Reeves et al. 2013), and, in species such as New Zealand sea lions, direct mortality from fisheries bycatch could result in population decline and future functional extinction (Chilvers 2012; Meyer et al. 2015). Similarly, adult NZFS bycatch in hoki (*Macruronus novaezelandiae*) trawls may be a factor in recent declines in pup numbers at colonies on the west of New Zealand's South Island (Roberts and Neale 2016). Another potential fishing related stressor for these colonies is competition for resources (Roberts and Neale 2016) as hoki is likely to be a seasonally important prey species for NZFS at these sites (Carey 1992). The idea of competition with fisheries potentially driving population level effects has been suggested for other species such as Stellar sea lions (*Eumetopias jubatus*) (Cornick et al. 2006), although this has been disputed (Hui et al. 2015).

Today, tourist disturbance represents another threat to pinnipeds (Boren et al. 2002). In addition to directly harmful interactions, such as tourists throwing stones at NZFS (Boren 2001), nuisance disturbances caused by human interactions can also be detrimental (Saltz et al. 2019). While not a direct threat, nuisance disturbances cause animals to allocate time and energy to assessing the disturbance, often at the expense of other behaviours. For example, NZFS alter their behaviour in response to land approaches, with reactions including avoidance, aggression and rest disruption (Boren et al. 2002). Particularly concerning for NZFS in popular tourist destinations such as Kaikōura is that the peak tourism period typically aligns with the NZFS' austral summer breeding season (Boren et al. 2002). Habituation is another possible outcome of repeated interactions with humans and is defined by Saltz et al. (2019) as the waning of response to a repeated, non-harmful stimulus. This too has been noted in areas where NZFS have been exposed to people for longer periods of time (Boren et al. 2002). While there may be immediate energy-saving benefits to not responding to human approaches (Mbise et al. 2020), habituation can decrease long-term fitness through reduced threat vigilance (Saltz et al. 2019; Mbise et al. 2020).

Anthropogenic coastal development has had significant consequences for pinniped species around the world. Habitat loss is considered one of the primary reasons for population declines in the critically endangered Mediterranean monk seal (*Monachus monachus*), which has largely abandoned its preferred open beach habitats and instead typically carries out parturition and pup rearing in caves (Johnson and Lavigne 1999). In caves, Mediterranean monk seal pups are vulnerable to mortality caused by impact against cave walls during storms and ocean swells (Gazo et al. 2000). Coastal development also led to the loss of colonies of South American sea lions (*Otaria flavescens*) on the Atlantic coast around Buenos Aires (Túnez et al. 2008). These colonies, which were not subject to largescale anthropogenic harvesting, were abundant until the second half of the nineteenth century, with their disappearance coinciding with significant human development in the area (Túnez et al. 2008). Human coastal development can also impact recolonisation processes. This has been demonstrated in New Zealand sea lions, which were extirpated from the New Zealand mainland by commercial hunting (Childerhouse and Gales 1998) but began breeding on South Island again in 1993 (McConkey et al. 2002). For this species, the presence of roads is the most important anthropogenic factor affecting the suitability of breeding habitat

(MacMillan et al. 2016). Around Kaikōura, road mortality is thought to be the most significant cause of mortality for NZFS pups over the age of 50 days (Boren et al. 2008).

Climate change is crucial in the assessment of threats to pinnipeds, with ongoing reductions in the extent, stability and duration of sea ice putting ice-associated species at particular risk (Kovacs et al. 2012). For example, ringed seals in Hudson Bay were in reduced body condition and had higher blubber cortisol levels, an indicator of stress, in 2013 compared to 1995, concurrent with longer ice-free periods (Ferguson et al. 2017). For terrestrially-breeding pinnipeds, there may be benefits to sea ice reductions, as these species may be able to colonise formerly ice-locked coastlines that were previously unavailable to them (Kovacs et al. 2012). However, within their current distributions, climate change is likely to have several direct effects on temperate and tropical pinniped species (Kovacs et al. 2012), including habitat loss due to increases in global sea levels (Kovacs et al. 2012), which could approach two metres by 2100, under very high emissions scenarios (Calvin et al. 2023). This could lead to habitat inundation, as has been projected in colonies of northern elephant seals in California (Funayama et al. 2013). McLean et al. (2018) note that rising sea levels will likely act synergistically with storm surges to affect colonies of Australian fur seals (*Arctocephalus pusillus doriferus*) in Bass Strait, and that, on average, by 2100, a 1-in-10-year storm will inundate a greater amount of habitat than a contemporary 1-in-100 year storm (McLean et al. 2018). This could cause consistently higher pup mortality or changes to population distribution (McLean et al. 2018). Rising ambient air temperatures will also likely have an effect (Kovacs et al. 2012), as large pinnipeds residing in temperate habitats can demonstrate significant thermal stress (Norris et al. 2010). Given that global warming is projected to exceed 2 °C by 2100 in all except the lowest emissions scenarios (Masson-Delmotte et al. 2021), it will become harder for pinnipeds to thermoregulate while ashore, with potentially severe consequences. For example, de Villiers and Roux (1992) showed that heat stress was the most significant environmental factor in the deaths of new-born Cape fur seals (*Arctocephalus pusillus pusillus*) in Namibia. Pinnipeds will also face changes to the marine environment stemming from climate change (Hoegh-Guldberg et al. 2007; Kovacs et al. 2012; Johnson and Lyman 2020; Calvin et al. 2023), however, a full discussion of such impacts is beyond the scope of this review.

The dense aggregations which many pinniped species form on land render them vulnerable to pathogens that spread through contact or proximity between individuals (Lynch et al. 2011). For example, hookworm (*Uncinaria* spp.) larvae, once expelled from one pup, can remain in the soil until they are picked up by another individual (Lyons et al. 2001). Hookworm infection is of particular concern to pups (Berón-Vera et al. 2004), as it can cause anaemia, haemorrhagic enteritis and, in severe cases, death (Duignan 2000). Berón-Vera et al. (2004) showed that, on average, 50% of 31 South American sea lion pups found dead at two different sites in Argentina had hookworm infections, and that prevalence increased with colony age and density. Proximity to anthropogenic activity has also been linked with disease incidence in pinnipeds. For example, Fulham et al. (2018) demonstrated that a colony of *Australian sea lions* (*Neophoca cinerea*) in South Australia located closer to human populations had higher incidences of human-associated *E. coli* phylotypes than a more isolated colony. Similarly, changes in antibody concentrations in the first three months of life among Galapagos sea lion pups (*Zalophus wollebaeki*) were negatively correlated with body condition measurements in a colony exposed to anthropogenic impacts on the environment, but not in a more isolated colony (Brock et al. 2013). This suggests that pups in the colony closer to anthropogenic activity were investing more into antigen development, potentially at the expense of body condition (Brock et al. 2013). Climate change means that pinnipeds, and particularly those at higher latitudes where temperatures are lower, will likely be exposed to diseases that they have not previously encountered (Harvell et al. 2002).

Pollution is another risk factor, with lipophilic substances such as polychlorinated biphenyls (PCBs) being particularly problematic to pinnipeds due to their cycles of feeding and fasting. This can mean individuals are exposed to concentrated amounts of the toxins while reliant upon breaking down their fat reserves, such as when mothers are fasting ashore as their pups wean (Kovacs et al. 2012). While pinniped populations closer to human settlements typically harbour greater pollution loads than those further away (Blanchet et al. 2021), a range of harmful pollutants have been detected in geographically isolated Antarctic pinniped species (Cipro et al. 2012). Marine plastic debris is also a documented threat to pinnipeds, particularly due to entanglement risks (Jepsen and de Bruyn 2019). In a review of 40 years of published pinniped entanglement records, Jepsen and de Bruyn (2019) showed that plastic fishing gear accounted for nearly all the pinniped entanglements on record. From analysing studies that mentioned both entangled and non-entangled pinnipeds, the authors calculated a global proportion of entangled individuals of 0.37 (Jepsen and de Bruyn 2019). NZFS in Kaikōura have particularly high entanglement rates, with plastics causing most incidents (Boren et al. 2006a). Boren et al. (2006a) suggest an increasing human population in the area, combined with NZFS recolonisation, could be behind high levels of entanglement. Plastics, and particularly microplastics, have also been found in pinniped guts (Hernandez-Milian et al. 2019), and it been hypothesised that such items could act as vectors for pollutants to enter marine mammals' bodies (Rochman 2015).

1.3 Impacts of marine oil spills on wildlife: an overview

Together with climate change, oil spills are among the most publicised anthropogenic impacts on the marine environment. This is, in large part, due to several high-profile oil spills that have had catastrophic impacts on proximate marine wildlife and habitats (Anderson 2002). These include infamous incidents such as the grounding of the *T/V Exxon Valdez* in Prince William Sound, Alaska, where the estimated loss of 2,650 sea otters (*Enhydra lutris*) came to symbolise the disaster (Garrott et al. 1993), the *Deepwater Horizon* disaster, where hundreds of thousands of birds (Haney et al. 2014), as well as many marine turtles and mammals (Williams et al. 2011) perished, and the sinking of the *MV Treasure*, in which 19,000 African penguins (*Spheniscus demersus*) were oiled, and a further 19,500 relocated to prevent their oiling (Wolfaardt et al. 2008).

The frequency of oil spills in both terrestrial and marine environments has increased in recent decades (Chilvers et al. 2021; Fan et al. 2021), with the contributions from different sources changing through time. Between 1970 – 2018, the contribution of oil spill incidents from general shipping has shown the highest and most continuous increase, while the number of spills originating from oil tankers has decreased in the last two decades (Chilvers et al. 2021), in large part thanks to improvements to tanker design (Chen et al. 2019). Marine wells and pipelines have shown a steady increase in incidents in recent decades (Chilvers et al. 2021). The worst oil spill to have taken place in New Zealand waters was the grounding of the *MV Rena* in the Bay of Plenty in 2011 (Schiel et al. 2016). While the volume of oil released (350 tonnes) was relatively small in comparison with the volumes released during events such as *Deepwater Horizon* or *Exxon Valdez*, there was a substantial impact on the local environment and wildlife (Schiel et al. 2016). As well as large volumes of oil washing up on the coastline (Schiel et al. 2016), nearly 2,500 seabirds, 24 terrestrial birds, 17 NZFS and four whales (Sievwright 2014) were oiled. A total of 383 little blue penguins (*Eudyptula minor*) were rescued and rehabilitated, with 90.6% surviving to be released back into the wild (Sievwright et al. 2019). Among the lessons from this event were the importance of baseline data to assess the impacts of spills, and the need to collate ecological data for areas where spills are likely to occur to inform future planning (Battershill et al. 2016).

A vast array of organisms has been impacted by oil spills in the marine environment, ranging from charismatic megafauna such as bottlenose dolphins (*Tursiops truncates*) (Pasamontes et al. 2017) to benthic organisms such as deep-sea corals (Girard and Fisher 2018). As exemplified in the *Deepwater Horizon* (Haney et al. 2014) and *MV Rena* (Sievwright et al. 2019) incidents, seabirds are often the most obviously impacted taxa following a marine oil spill (Fraser et al. 2022). Seabirds are at particular risk due to the amount of time they spend at the surface, giving them a high chance of encountering a slick. The most prominent risk to birds in oil spill incidents comes from feather fouling, and, given the important roles that feathers play in insulation, water repellence and flight, the results of oiling are often fatal (Leighton 1993). In addition, as well as ingesting or inhaling oil directly from the marine environment, preening of feathers can also cause oil ingestion (Burger 1997; King et al. 2021), potentially leading to endocrine disruption. Compared to birds and other well-studied species such as sea otters (Garrott et al. 1993; Bodkin et al. 2002; Jessup et al. 2012), relatively little is known about the effects of oil spills on pinnipeds (Helm et al. 2015). However, we do know that fur seals are at greater risk of some of the deleterious impacts of oil spills than sea lions, phocids and the walrus (*Odobenus rosmarus*) (Helm et al. 2015). In part, this is because, like sea otters, fur seals rely on fur rather than blubber to insulate themselves, meaning that fouling can inhibit thermoregulation (Mearns et al. 1999; Helm et al. 2015). In oiled sea otters, the inability to stay warm while diving leads to shortened foraging dives and shallower dive profiles (Helm et al. 2015), impacting their foraging ability. Similarly, oiled fur seals haul out more frequently to stay warm (Ziccardi et al. 2015). The dense aggregations formed by fur seals on land, particularly during breeding seasons, also put these species at greater risk than more solitary pinnipeds. Oil spills near breeding colonies may have particularly severe consequences, as exemplified by the spill from the tanker *San Jorge* off Uruguay in 1997, where 4,738 South American fur seal (*Arctocephalus australis*) pup carcasses were collected following the incident, although the actual mortality was presumed to have been higher (Mearns et al. 1999). Similarly, all 39 pups observed at a small colony of NZFS on Hood Island impacted by an oil spill from the 1991 sinking of the *Sanko Harvest* off West Australia were found to have been heavily oiled, while a smaller number at a larger colony on Seal Rocks were oiled in the same incident (Gales 1991). As otariid pups are mostly confined to land until they wean and have longer weaning periods than phocids (Atkinson et al. 2011), they have limited means of escape if oil washes ashore at breeding sites (Jernelv 2010). Because pups cannot fully sustain themselves until weaned, spill induced disruptions to regular feeding by their mothers can lead to starvation (Mearns et al. 1999). As well as their high vulnerability to oil spills, fur seals are also difficult to rescue and rehabilitate. For adults and sub-adults, their size and potentially aggressive behaviour can make them difficult to handle (Northwest Area Committee 2019), and even smaller individuals require large numbers of personnel to adequately handle and care for them during transportation, housing and cleaning (Ziccardi et al. 2015). For example, effectively cleaning dense fur seal fur typically involves teams of four to six people, while team sizes for other species typically number two to three (Ziccardi et al. 2015).

There are three principal aspects to an oiled wildlife response: 1). Stopping the oil from reaching wildlife, 2). Removing wildlife from the path of the oil, and 3). Capturing, cleaning, rehabilitating and releasing the wildlife (Wolfaardt et al. 2008; Nijkamp et al. 2014; Chilvers and Battley 2019; Chilvers and McClelland 2023; Chilvers and Ruoppolo 2023). As preventing wildlife oiling is the optimum response option, planning and preparedness is key to enabling rapid mobilisation of personnel and resources (Moller et al. 2003; Clumpner and Callahan 2014; Chilvers et al. 2021; Yeung et al. 2021; Chilvers and McClelland 2023).

Among the first steps in preparing a wildlife response plan is knowing which species are likely to be in a region that could be impacted by an oil spill. Importantly, this should consider variable factors like how a species' local distribution and abundance may change with season and environmental drivers (Chilvers and Battley 2019; Chilvers and Ruoppolo 2023). To prioritise, during an oiled wildlife rescue and rehabilitation response, such plans also need to consider varying degrees of rarity, vulnerability to oil spills in different species, how they are likely to respond to capture and rehabilitation, and the different resources needed (Chilvers and Battley 2019). Once an oil spill has occurred, a variety of tools and techniques have historically been deployed to attempt to minimise the impacts on marine wildlife and habitats. These include mechanical measures, such as containment booms, skimmers and sorbents; chemical methods such as dispersants or emulsion breakers, in situ burning and bioremediation, which involves introducing agents to expedite natural degradation processes (Mullin and Champ 2003; Ventikos et al. 2004). Which measures, if any, are deployed will depend on a range of factors, including the ocean conditions, the velocity and viscosity of the oil spill (Ventikos et al. 2004), and the potential impacts to the environment and wildlife (Mullin and Champ 2003).

Previous responses to oiled fur seals have varied in scope. During the *San Jorge* incident, rehabilitation of oiled South American fur seal pups was discouraged from the outset, as it was deemed logistically impossible for thousands of pups to be cleaned on a remote island, and survival was predicted to be very low even if this was attempted (Mearns et al. 1999). Despite this, 41 pups were transferred by nongovernmental organisations for cleaning, with only three surviving into late March, following the incident on February 8th, 1997 (Mearns et al. 1999). By contrast, all the observed contaminated pups on Hood Island (n=39) and 172 pups on Seal Rocks (both contaminated and not) were captured and restrained in pens on the respective islands following the *Sanko Harvest* spill (Gales 1991). In this response, a nonsolvent, biodegradable, non-ionic detergent and a hydrocarbon solvent were used to clean oiled pups, and colony habitats were also cleaned prior to pup release (Gales 1991). Mortality of pups at Hood Island was estimated at between 13 – 33%, but it was not possible to determine what extent of the mortality was attributable directly to oiling, as opposed to clean-up related disturbance or unrelated causes (Gales 1991). Meaningful estimates of mortality from Seal Rocks were not attempted (Gales 1991).

One of the most challenging elements of responding to an oil spill impacting marine mammals is establishing and maintaining adequate treatment and rehabilitation facilities (Ziccardi et al. 2015). In the National Oceanic and Atmospheric Administration (NOAA) guidelines for responding to oiled pinnipeds and cetaceans in the United States, it is noted that facilities will often need to be improved or retrofitted to deal with oiled marine mammals, as spills will not necessarily occur near permanent rehabilitation centres (Ziccardi et al. 2015). While this document states that no model facility exists that can cater for all age classes and species of marine mammal, it does list the features that an ideal facility should contain (Ziccardi et al. 2015). The importance of adequate facilities was demonstrated during the *Sanko Harvest* response, when six pups died of asphyxiation in a crush in a capture pen that contained a large group of animals and was left unattended overnight (Gales 1991). In this response, pens on the affected islands were used as opposed to transporting pups to rehabilitation facilities elsewhere, despite public and media calls for the latter approach, as it was believed that transportation would have resulted in higher pup mortality (Gales 1991).

There are no permanent facilities in New Zealand capable of dealing with substantial numbers of oiled NZFS, with the current permit for dealing with oiled seal species in New Zealand (Permit number: 86142-MAR) listing Auckland Zoo as the primary location for seals that cannot be treated *in situ*. There is also an insufficient number of trained personnel and a lack of the equipment that would be needed to conduct a large scale NZFS rehabilitation operation (Chilvers 2017). As such, and particularly given the likelihood of high public interest in the welfare of this charismatic species (Chilvers 2017), pre-emptive measures that stop NZFS from becoming oiled in the first place represent the best course of action during an oil spill that threatens this species. To achieve this, the distribution and abundance of the species in the impacted area must be known.

1.4 Road impacts on wildlife: an overview

The most obvious impact of roads on wildlife is mortality resulting from collisions with vehicles. Decreasing local populations proximal to roads have been demonstrated in a wide range of wildlife, including reptiles (Beaudry et al. 2010), insects (Baxter-Gilbert et al. 2015), mammals (Ramp and Ben-Ami 2006) and birds (Borda-de-Água et al. 2014) - with collisions contributing substantially to these declines (Santos et al. 2018). A significant volume of research has been dedicated to locating and attempting to understand what causes roadkill hotspots (e.g. Seiler 2005; Langen et al. 2012; Bíl et al. 2019). There is also a human dimension to these incidents. For example, between 2003–13, in São Paulo, Brazil there were an average of 2,611 collisions per year involving animals, 18.5% of which resulted in human injuries or fatalities, at an estimated annual cost of over US$25 million per year (Abra et al. 2019).

Roads also impact the quality and quantity of habitat available to wildlife (Augé et al. 2012; Whittington et al. 2019; Zhang et al. 2020). These effects include vegetation clearance (Toulec et al. 2020), increases in local pollution, including light and noise (Koemle et al. 2018) air-polluting gas emissions (Huang et al. 2009), and the run-off of toxins into the local environment (Kunz et al. 2022). Ghadirian et al. (2019) demonstrated how road impacts go beyond the structure itself in a study of Persian gazelle (*Gazella subgutarosa*), finding that road noise impacted up to 8,308 hectares (ha) of the road's surroundings, even though the infrastructure itself only resulted in the direct loss of 132 ha of habitat.

Roads create barrier effects resulting either from animals experiencing physical difficulties in crossing the road, for example where steep verges exist (Marsh et al. 2005), or behavioural changes where animals avoid roads (Northrup et al. 2012). Sawyer et al. (2012) found that roads were contributing to bottlenecks in the migration routes of mule deer (*Odocoileus hemionus*) and pronghorn (*Antilocapra americana*) populations, reducing the effectiveness of these routes. Similarly, Keller and Bender (2007) found that bighorn sheep (*Ovis canadensis*) required more attempts to cross a road to reach an important habitat area in Rocky Mountain National Park when greater numbers of vehicles and visitors were present. Sometimes, the habitats that animals are precluded from accessing are used for specific biological processes, meaning that impeded access could have detrimental population effects (Keller and Bender 2007). Roads can also disrupt gene flow, potentially altering population structures (Jaeger and Fahrig 2004) and leaving small sink populations at greater risk of local extinction (Benson et al. 2016). For example, microsatellite data from moose (*Alces alces*) in Alaska showed evidence of a genetic subdivision between populations that corresponded chronologically to the construction of highway infrastructure, and the associated barrier effect (Wilson et al. 2015b). While there is some evidence for reduced genetic diversity in the face of barrier-induced population isolation (Epps et al. 2005; Delaney et al. 2010) other studies have noted no such change, even when a barrier effect was evident (Wilson et al. 2015b).

Although roads may impede movement in some species, they can expedite it in others, with predators known to use roads to quickly access different habitats for hunting (Hradsky et al. 2017). Roads can also provide access for humans to remote areas and bring them into increased contact with wildlife (Bernstein et al. 2022), potentially facilitating deleterious interactions such as poaching (Soofi et al. 2022) and habitat destruction (He et al. 2019). A less overtly harmful, but nonetheless important, impact of access provision to humans relates to eco-tourism, and the associated threats described in Section 1.2.6.

There have been numerous studies investigating methods for mitigating roads' impacts on wildlife (Jaeger and Fahrig 2004; Taylor and Goldingay 2010; Huijser et al. 2016; Spanowicz et al. 2020). Most involve controlling wildlife access to roads (Dean et al. 2019), with fencing being a frequently employed strategy (Jaeger and Fahrig 2004). While road fencing has been effective in reducing road mortality (Spanowicz et al. 2020), this approach can have drawbacks, especially when used in isolation. For example, it is often not possible to fence entire roads (Dean et al. 2019), which can create 'fence-end effects', whereby animal crossings and collisions are concentrated where the fencing finishes (Huijser et al. 2016). Huijser et al. (2016) found fence sections under five kilometres in length reduced large mammal-vehicle collisions by 52.7%, whereas fences longer than five kilometres showed over an 80% reduction in large mammal-vehicle collisions. In addition, the effectiveness of shorter fence sections was more variable than that of longer fence sections. The efficacy of a fence can also depend on the species it is trying to protect and the traffic volume (Jaeger and Fahrig 2004). Jaeger and Fahring (2004) found that above a species-specific road mortality threshold, a fence is always beneficial to population persistence, while below a certain threshold of species-specific road mortality, fencing is always detrimental. Between these thresholds, a fence's impact was linked to the population's degree of road avoidance (Jaeger and Fahrig 2004). Jaeger and Fahring (2004) recommend that fences should be used when the volume of traffic on a given road is so great that animals are almost never able to cross successfully, or in cases where the population is declining, and road mortality is a known factor in this decline.

Even when beneficial to population persistence, Jaeger and Fahring (2004) believe fencing should only ever be an interim measure, to reduce the road's ongoing negative impacts on the population, while a more permanent solution is sought. This is because fencing exacerbates the barrier effect (Jaeger and Fahrig 2004). Thus, it is increasingly common for crossing structures to be built to allow animals to pass safely and reduce the risks to motorists (Denneboom et al. 2021). Such structures range from viaducts and underpasses (Denneboom et al. 2021) to box culverts and concrete pipes (González-Gallina et al. 2018). For such structures to be effective, sites that offer functional connectivity for the species in question must be identified (Zeller et al. 2020). Additionally, factors that may influence whether wildlife choose to use a given crossing structure, for example, whether the structure is also used by humans, should be considered (González-Gallina et al. 2018).

1.5 New Zealand fur seal in the Canterbury region

Canterbury is a ca. 45, 300 km^2 central-eastern region of New Zealand's South Island (Figure 1.1). Ōtautahi/Christchurch is the region's largest city. This study will mainly focus on NZFS populations in two distinct areas of Canterbury – Kaikōura and Banks Peninsula. Below are descriptions of both regions, summaries of previous NZFS studies in each, and an introduction to two distinct threats to the species in each location.

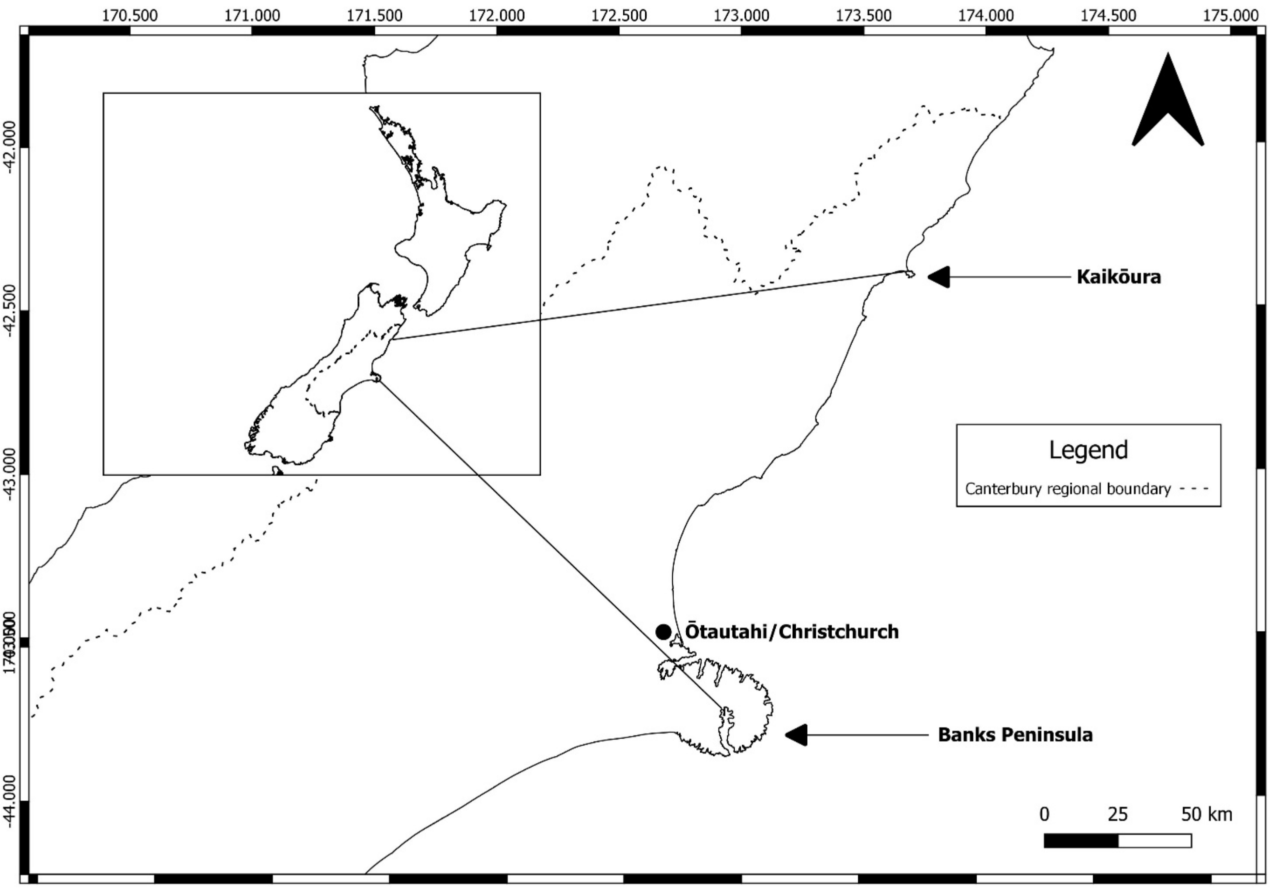


Figure 1.1 The location of the study areas in relation to the Canterbury region and New Zealand

1.5.1 Kaikōura

Kaikōura township is ca. 182 km by road from Ōtautahi/Christchurch (Figure 1.1), and the Kaikōura district is home to substantial marine wildlife populations, in large part due to the Kaikōura submarine canyon - the most productive non-chemosynthetic habitat recorded in the deep sea (De Leo et al. 2010). The Kaikōura Canyon is an important feeding area for a variety of marine mammal species such as dusky dolphins (*Lagenorhynchus obscurus*) (Benoit-Bird et al. 2004) and sperm whales (*Physeter macrocephalus*) (Guerra et al. 2017, 2020) as well as a plethora of seabirds (Stonehouse 1965). The canyon head is located ~500 m from shore on the south side of Kaikōura Peninsula, and the trench itself is 60 km long and up to 1,200 m deep (Lewis and Barnes 1999). Benoit-Bird et al. (2004) found that the deep scattering layer, where dusky dolphins feed on squid and myctophids, is within 150 m of the surface between 1900 to 0530 hours, and within 50 m of the surface between 2300 and 0100 hours. Myctophids and squid are known prey of NZFS in Kaikōura (Emami-Khoyi et al. 2016b), and, given the depths to which NZFS are capable of diving (Mattlin et al. 1998), Boren (2005) suggests that the food sources offered by the Kaikōura Canyon could be accessible to NZFS all year round.

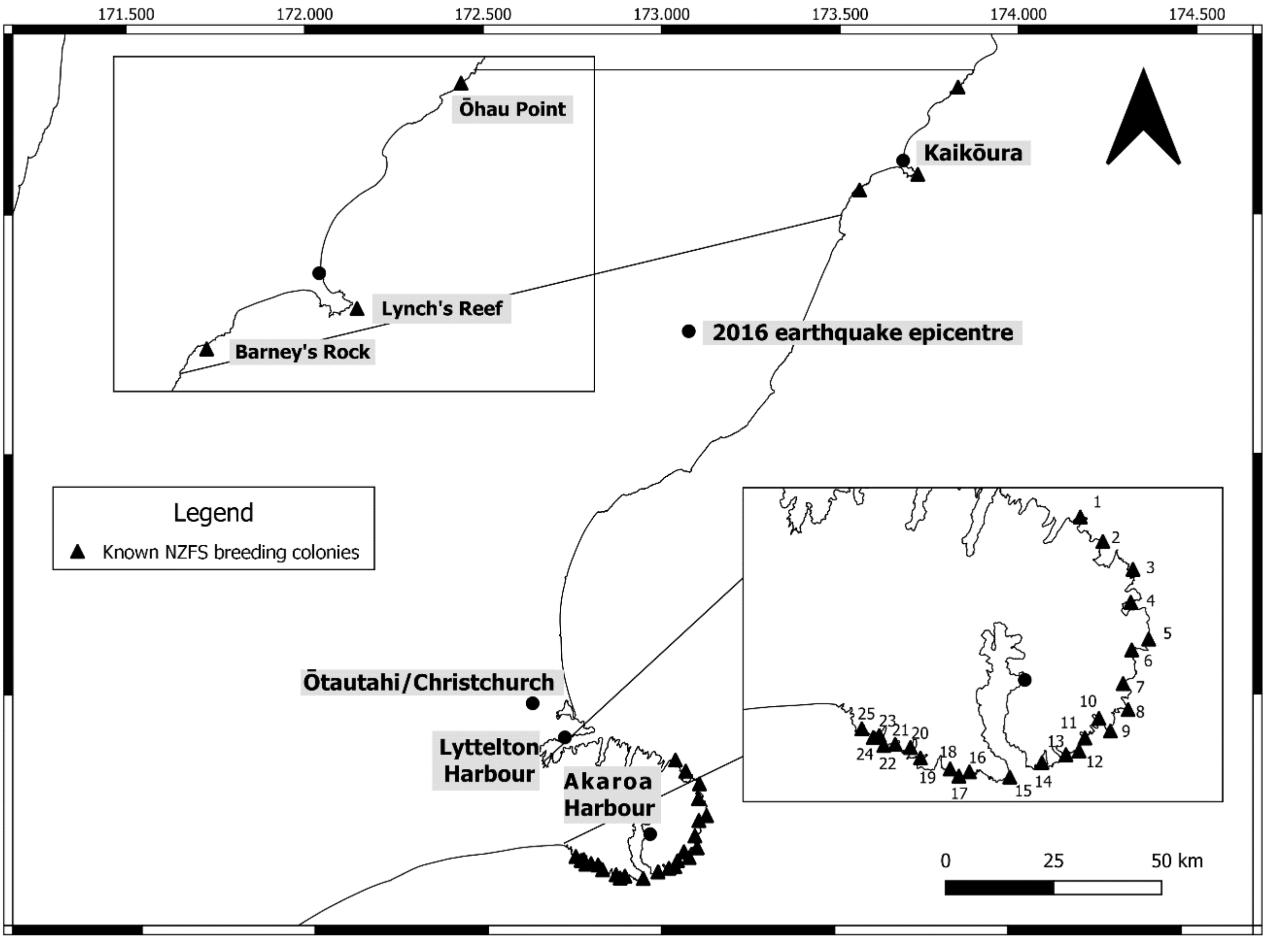


Figure 1.2 Important locations within the study area, including known New Zealand fur seal breeding colonies. The Banks Peninsula colonies are numbered. 1. Long Lookout, 2. West Head, 3. Ducksfoot Bay, 4. Le Bons Bay, 5. East Head, 6. Hickory Bay, 7. Goughs Bay, 8. Goat Point, 9. Pompey's Pillar, 10. Otanerito Bay, 11. Short Reef Point, 12. Redcliffe Head, 13. Flea Bay, 14. Damons Bay, 15. Waihuakina Bay, 16. Rocky Nook, 17. Whakamoa Bay, 18. Island Bay, 19. Horseshoe Bay, 20. Peraki Bay, 21. Robin Hood Bay, 22. Hell Gate, 23 Te Oka Bay, 24. Boaz, 25. Murray's Mistake. Baird (2011) includes two other colonies not depicted here because the names given did not correspond to locations that could be identified on a map: North Head and Pukalolo Bay

Post-exploitation, substantial numbers of NZFS were first recorded in Kaikōura in 1958 (Street 1964). Stirling (1971) reported a maximum of 800 individuals on the Kaikōura Peninsula in winter, and between 200-300 in summer, describing the population as a non-breeding colony. By 1990, pups were being born at three sites along the Greater Kaikōura coastline, at Ōhau Point, north of the township, on the Kaikōura Peninsula, and at Barney's Rock, offshore from Kaikōura's south coast (Boren et al. 2006b) (Figure 1.2). From 1990 – 2005, the number of NZFS recorded during helicopter surveys increased from around 1,200 to over 3,000 between the Clarence River in the north and the Waiau River in the south (Boren et al. 2006b).

Studies of population growth in Kaikōura's NZFS have focussed on the Ōhau Point colony and, to a lesser extent, Lynch's Reef on the Kaikōura peninsula. Approximately 50 pups were recorded at Ōhau Point in 1996, which had increased to 110 by 1998, and between 6 – 8 pups at Lynch's Reef over the same period (Barton et al. 1998). Between 2002 – 2005, Boren et al. (2006b) conducted pup abundance studies at both colonies. They found pup production at Ōhau Point to be growing exponentially at a rate of 32% annually, with a Peterson estimator of almost 600 pups in 2005. The number of pups produced at Lynch's Reef was far smaller, between 8 – 12 per year through the study, which may have been the site's spatial carrying capacity (Boren et al. 2006b). In 2015, a mark-recapture study at Ōhau Point produced a Peterson estimator of 2,471 pups (Gooday 2016). Shortly after this study a magnitude 7.8 earthquake struck the Kaikōura region (Stringer et al. 2021), and no mark-recapture studies of the local NZFS population have been conducted since. Direct counts by Department of Conservation (DOC) staff in winter 2021 found that NZFS were breeding at several sites along the Greater Kaikōura Coastline in addition to those shown in Figure 1.2 (J. Weir and L. Boren 2021, pers. comm.).

1.5.1.1 The 2016 Kaikōura earthquake

Just after midnight on November 14th, 2016, a magnitude 7.8 earthquake struck 15 km northeast of Culverden (Furlong and Herman 2017), a small town situated roughly 105 km south-west of Kaikōura. After the main shock, the bulk of the aftershock activity was felt around the Kaikōura and Seddon regions (Harte 2019).

The earthquake caused substantial and varied impacts on Kaikōura's marine and terrestrial environments. Tsunami waves arrived in Kaikōura roughly 30 minutes after the initial rupture, with wave heights reaching up to 2.5 m along the coast (Duputel and Rivera 2017). In addition, over 10,000 landslides were triggered over roughly 10,000 km^2 (Massey et al. 2018; Guo et al. 2019), with most concentrated over 3,600 km^2 (Massey et al. 2018). At Ōhau Point, over 100,000 m^3 of debris fell onto the coastline and important road and railway infrastructure below (Barrett and Hayes 2019). These landslides had significant impacts on the economically important South Kaikōura Transport Corridor, leaving the town cut off from coastal road access (Stringer et al. 2021). As such, a large-scale reconstruction effort was undertaken to clear the roads and repair the damage. Underwater, the earthquake triggered extensive landslides in the Kaikōura Canyon, resulting in a powerful flushing event, which led to an estimated loss of 39×10^6 kg of benthic biomass from the canyon ecosystem (Mountjoy et al. 2018). Given the importance of benthic communities to canyon food webs (De Leo et al. 2010), this is likely to have impacted predators foraging in both pelagic and demersal zones (Guerra et al. 2020).

Little is known about the impacts of earthquakes on mobile marine species (Guerra et al. 2020). Except for Guerra et al.'s (2020) study on Kaikōura's sperm whales and Gallo-Reynoso et al.'s (2011) observations of a fin whale (*Balaenoptera physalus*) following an earthquake, the impacts on marine mammals have not been documented. It is not known how many NZFS died as a direct result of the November 2016 Kaikōura earthquake, although unpublished data from the North Canterbury Transport Infrastructure Recovery (NCTIR) group indicates that mortality did occur. In colonially breeding species, the timing of natural disasters may impact different population cohorts to different extents. For example, Chilvers and Hiscock (2019) recorded mortality biased towards adult male erect-crested penguins (*Eudyptes sclateri*) on the Antipodes Islands following a storm and landslides in 2014. This was likely because male erect-crested penguins are solely responsible for chick guarding in the three to four weeks post-hatching (Mattern and Wilson 2018) while females forage at sea. By contrast, there is unlikely to have been such clear sex segregation among the estimated 40,000 Hutton's shearwaters (*Puffinus huttoni*) killed in the 2016 Kaikōura earthquake, as parents in this species alternate incubation duties weekly (Cuthbert 2019). The date (November 14th) of the 2016 Kaikōura earthquake makes it unlikely that many, if any, new-born NZFS pups were present in the colonies. Boren (2005) recorded the first pup birth at Ōhau Point on November 26th in 2002, November 13th in 2003 and November 17th in 2004. However, deceased pups were discovered under rubble in 2017, potentially killed during subsequent slips (NCTIR, unpublished data). Boren's (2005) study of colony dynamics at Ōhau Point indicate that both adult males and females, including pregnant females, were likely to have been ashore when the earthquake happened. Indeed, the number of females hauling out at the colony peaked on November 14th in 2003, and November 13th in 2004 (Boren 2005). However, the fact that the earthquake occurred around midnight (Furlong and Herman 2017), may have reduced overall mortality because NZFS are primarily nocturnal foragers (Page et al. 2005). That said, stressful events can lead pregnant fur seals to spontaneously abort foetuses (Osiecka et al. 2020), meaning the earthquake could have indirectly reduced pup production in the subsequent season. An unknown number of adults and pups were also killed during the road reconstruction effort (NCTIR, unpublished data).

Key terrestrial habitat areas for NZFS were among those impacted by the earthquake, particularly the Ōhau Point breeding colony (Barrett and Hayes 2019) and the associated waterfall stream which had been a congregating site for pups (Acevedo-Gutiérrez et al. 2011). As well as rockslides, the earthquake caused uplift of between zero and six metres at different locations along the coastline (Alestra et al. 2019). At Goose Bay, on the south coast of Kaikōura, this uplift was 1.6 m, around the Kaikōura Peninsula it was 1.1 m, while at Ōhau Point the uplift was between 2 – 3 m (Hay 2020). At Ōhau Point, earthquake induced uplift, and the subsequent road reconstruction which narrowed the original colony to approximately one-third of its former depth, resulted in the loss of many large caves, crevices and rock pools (Gooday and Goldstien 2018). Using Taylor et al.'s (1995) NZFS habitat assessment criteria, Gooday and Goldstien (2018) downgraded Ōhau Point's suitability as a breeding habitat from nine (the highest possible), pre-earthquake, to six. However, pups were born at Ōhau Point in the breeding season immediately following the earthquake (Gooday and Goldstien 2018), indicating that NZFS may continue to use habitats of reduced quality in the aftermath of natural disasters. This phenomenon has been recorded in philopatric aquatic (Lai et al. 2007) and terrestrial species (Dudley et al. 2022) following natural disasters.

By comparison, the habitat available to NZFS at Lynch's Reef on the Kaikōura Peninsula increased due to uplift, providing more area to a colony where growth was thought to be space limited (Boren et al. 2006b). Post-earthquake, more cows were noticed settling at the Kaikōura peninsula (O. Gooday, pers. comm.). Habitat enlargements following natural disasters have also been observed with northern elephant seals at Point Reyes, California, where landslides deposited additional sediment at the rear of their beach habitat (Funayama et al. 2013). However, for NZFS at Lynch's Reef, the uplift also resulted in the loss of a deep channel that had offered the site some degree of protection from tourist disturbance (Gooday and Goldstien 2018).

The 2016 Kaikōura earthquake also likely temporarily altered the quality of the Kaikōura Canyon as a foraging habitat (Guerra et al. 2020). Guerra et al. (2020) recorded changes in foraging behaviours and the distribution of high use areas among Kaikōura's sperm whales, including greater preference for sites further offshore. This could be indicative of whales expending more energy to capture prey, while the altered distribution patterns could demonstrate use of suboptimal habitats (Guerra et al. 2020). However, after a period of a year, most of the recorded sperm whale foraging parameters had returned to pre-earthquake status, indicating at least partial recovery (Guerra et al. 2020).

Following the earthquake, an average of 766 pups were counted at Ōhau Point from four helicopter counts between March 15th and April 2017 (Gooday and Goldstien 2018). Although these counts cannot be directly compared with the most recent pre-earthquake mark-recapture estimate of 2,471 pups (Gooday 2016), the 2017 figure represents a substantial decrease (Gooday and Goldstien 2018). In addition to mortality and potential stress-induced foetal abortion, another potential reason for reduced NZFS numbers at Ōhau Point could be site abandonment. Large-scale disturbances can induce surviving individuals to leave formerly occupied sites (Reiley et al. 2013). Indeed, colonies immediately to the south and the north of Ōhau Point (Half Moon Bay and Paparoa Point), saw increases in NZFS numbers following the earthquake, although this was partially due to human relocation of animals (Gooday and Goldstien 2018). NZFS abundance at Half Moon Bay increased from 200 recorded in a ground count in May 2011 to 1,265 counted from a helicopter in April 2017, while, at Paparoa Point, numbers increased from 420 in September 2002 (ground count) to 1,573 in April 2017 (helicopter count) (Gooday and Goldstien 2018). Both pups and adults were relocated from Ōhau Point to other sections of the coastline, including Paparoa Point, approximately 2 km north, during the post-earthquake reconstruction (Gooday and

Goldstien 2018). However, the number of pups at Paparoa Point subsequently decreased (Gooday and Goldstien 2018).

1.5.1.2 Kaikōura's NZFS and State Highway 1

State Highway 1 (SH1) runs the length of New Zealand and is a key piece of national transport infrastructure. In the Kaikōura region, it experiences an annual average daily traffic (AADT) volume of over 2,500 vehicles, of which roughly 10% are heavy good vehicles (gross vehicle mass of over 3.5 tonnes) (Aghababaei et al. 2020). It also abuts several NZFS breeding sites.

Between 1996 – 2005, DOC callout data showed 120 NZFS were reported on SH1, an annual average of 12 (Boren et al. 2008). Over 40% of these were killed by vehicles (Boren et al. 2008). Between 1996 – 2005, the average number of pups killed on the road per year was estimated to be between 2.4 ± 1.4% of those born each year (Boren et al. 2008). While mitigation measures were implemented to exclude NZFS from SH1, including a crash barrier extending the full length of the pre-earthquake Ōhau Point colony, NZFS were still able to access the road (Boren et al. 2008). As the Kaikōura NZFS population was expected to continue growing, Boren et al. (2008) predicted that increasing numbers of NZFS would access SH1 in the future.

1.5.2 Banks Peninsula

Banks Peninsula, a peninsula of ca. 1,150 km^2, lies southeast of Ōtautahi/Christchurch (Figure 1.1). It features many bays and inlets, and two important ports, Lyttelton and Akaroa Harbour (Carome et al. 2022). The region has been described as an ecological hotspot for marine species travelling through New Zealand waters due to its geographic location (Banks et al. 2002).

Wilson (1981) estimated a range of 50 – 100 NZFS around Banks Peninsula in 1973, mostly non-breeding males and immature individuals. However, four pups and one female were reported at Horseshoe Bay (Wilson 1981). Ryan et al. (1997) described a total of 15 colonies around the peninsula, although only five (Horseshoe Bay, Flea Bay, Pompey's Pillar, Goat Point South and Ducksfoot Bay) were described as breeding sites. Boren (2005) reported that the Department of Conservation received notification that three pups were born also born at Te Oka Bay in 1997.

By the time of Boren et al. (2006b) two colonies identified by Ryan et al. (1997) as non-breeding colonies (Whakamoa Bay and Island Bay) had become breeding sites, and breeding had become fully established at Te Oka Bay, with 42 pups born there in 2001. Boren (2005) suggested that pupping at Whakamoa Bay, Island Bay and Te Oka Bay represented spill over from nearby Horseshoe Bay. In the final year of their study, Boren et al. (2006b) estimated that 300 pups were born at Te Oka Bay, and a similar number at Horseshoe Bay. While the Horseshoe Bay colony appeared to be relatively stable, with no great increases in pup production since the late 1990s (Boren 2005), Te Oka Bay was growing rapidly at 47% p.a.. By 2011, there were breeding colonies in most of the southern and eastern bays of Banks Peninsula (Allum and Maddigan 2012) with Baird (2011) listing a total of 25 pupping sites (Figure 1.2). The most recent NZFS population estimate for Banks Peninsula was estimated at 5,136 individuals (M. Morrissey, *unpublished data*) from aerial surveys in 2007. However, the results of these surveys should be treated with some caution due to uncertainties regarding their timing, precise methodologies and geographic extent. Emami-Khoyi et al. (2016a) conducted genetic sampling on Banks Peninsula, taking samples from the 10 "major" colonies, all consisting of more than 50 individuals. They suggested the NZFS populations on Banks Peninsula are in the maturity stage of Roux' (1987) recolonisation framework.

1.5.2.1 Oil spill risk on Banks Peninsula

Banks Peninsula contains two important harbours. Lyttelton is the major hub port on South Island, and one of the busiest in the country (Inglis et al. 2008), with 889 large vessels requiring a pilot boat arriving in 2023 (Lyttelton Port Company, pers. comm.). Akaroa is not a commercial port, but is a popular tourist destination, particularly among cruise tourists (Wilson et al. 2015a; Hussain and Fusté-Forné 2021).

Oil spill risks for these two ports, and thus the surrounding areas, are likely to grow over the coming years. The recent closure of the Marsden Point oil refinery means that New Zealand is now totally reliant on imports of oil (Wilson et al. 2023), resulting in a greater number of oil transportation vessels in New Zealand waters. Increased transportation, and thus consumption, of fuel oil in recent decades has driven the global increase in oil spills in both marine and terrestrial environments and combined, oil tankers and general shipping have accounted for 70% of spills reported between 1970 – 2018 (Chilvers et al. 2021). As such, ocean and coastal wildlife managers in New Zealand should be aware of the potential increased risk stemming from these changes and put in place rigorous and practical wildlife response plans. In Lyttelton Harbour, the risk of oil spills is mainly from tankers, with 1,056,663t of fuel imported by the Lyttelton Port Company in 2023 (Lyttelton Port Company 2023), although the harbour does also experience a substantial amount of traffic from cruise ships, large container ships and log carrying vessels, and vessel traffic of any kind is a potential source of oil spills (Chilvers et al. 2021).

In Akaroa Harbour, the oil spill risk comes predominantly from cruise ships. Prior to the COVID-19 pandemic, cruise ship tourism was among the fastest growing segments of global tourism, with a 24% increase in cruise ship passengers recorded in New Zealand between 2018 and 2019 (Carome et al. 2022). Should similar trends occur post-pandemic, this will add to the number of ships in New Zealand waters. While large scale oil spills from cruise ships are relatively rare, there have been a number of high-profile events, such as the sinking of the MS Explorer in Antarctica in November 2007 (Stewart and Draper 2008) and the Costa Concordia wreck off Italy in January 2012 (Frodella et al. 2023).

In September 2023, during the preparation of this thesis, a 25-metre fishing vessel, *Austro Carina*, ran aground near Red Bay on the southeastern side of Banks Peninsula carrying 10,000 litres of diesel and 400 litres of hydraulic oil (Radio New Zealand 2023a). As well as being near known NZFS breeding sites, Shell Bay is a habitat for endangered species such as the yellow-eyed penguin (*Megadyptes antipodes*) and spotted shag (*Stictocarbo punctatus*) (Radio New Zealand 2023b). While there were no reports of oiled wildlife, this incident underscores the need for robust oil spill response plans, underpinned by contemporary ecological data, to safeguard NZFS and other marine species found around Banks Peninsula.

1.6 Thesis structure and aims

This thesis is organised into six chapters: 4 data chapters, a general introduction chapter and a final discussion chapter. The data chapters have been written in publication format, and have either been published (Chapters 2 and 3) have been accepted but not yet published (Chapter 5) or are currently under review (Chapter 4).

Chapter 1: General Introduction

This chapter summarises previous research on NZFS biology and ecology, as well as the main natural and anthropogenic risks to the species throughout its range. It also provides outlines of the core methodologies employed in the data chapters (Chapters 2,3, 4 and 5), describes the study areas and overviews past NZFS population studies specific to these locations.

Chapter 2: Earthquake impacts on a protected pinniped in New Zealand

This paper updates our understanding of NZFS abundance and the distribution of breeding colonies of NZFS along the Greater Kaikōura Coastline. As well as the broader context of post-sealing population recovery, this paper provides the first assessments of two earthquake affected sites - Ōhau Point and Lynch's Reef - since the earthquake, as well as sites that have not been previously documented. From these assessments, an update is provided on the population of NZFS along the Greater Kaikōura Coastline as a baseline for future monitoring, and suggestions are made with regards to how contemporary conservation and management measures could be updated.

Chapter 3: Post-earthquake highway reconstruction: Impacts and mitigation opportunities for a New Zealand pinniped population

This paper provides an understanding of the risks posed to Kaikōura's NZFS population by a major road, State Highway 1 (SH1).

This builds on previous research conducted prior to the 2016 earthquake to provide an updated account of the current impacts of SH1 on the local NZFS population. Specifically, it identifies the locations for hotspots of NZFS incidents, examines whether these locations have changed through time, and models the spatial, temporal and environmental factors that contribute to the timing and location of such incidents. The intention is to provide information that will be useful to managers seeking to mitigate this issue by highlighting when and where NZFS are likely to access SH1, so that management measures can be enacted.

Chapter 4: Planning for a pinniped response during a marine oil spill

This paper provides the first assessment of the distribution and abundance of Banks Peninsula's NZFS in nearly 20 years, with the aim of informing oil spill response plans.

New Zealand's marine oil spill risk has increased due to the closure of the Marsden Point refinery, meaning that the country now relies entirely on imported oil, as well as the recovery of cruise tourism following the COVID-19 pandemic. NZFS on Banks Peninsula are proximate to two ports – Akaroa and Lyttelton – however, there are currently no contemporary data on the distribution and abundance of the local NZFS population, which would impede efforts to protect them from oil.

Chapter 5: Towards a total abundance estimate of New Zealand fur seal in New Zealand

This paper provides the first empirical update to our understanding of NZFS abundance in New Zealand waters in ca. 50 years. This is achieved through a combination of using the most recently available data from known breeding colonies and stage-structured matrix models, to estimate contemporary abundance.

In addition to the population figures themselves, this paper aims to highlight the inconsistencies in NZFS monitoring throughout most of New Zealand and examines some of the benefits that greater knowledge of NZFS population parameters would provide to managers.

Chapter 6: Discussion and Conclusion

Chapter 6 summarises the main findings of the thesis and discusses the implications of these findings for the conservation of NZFS and other otariid species, both in New Zealand and around the world. It concludes with suggestions for further research and management of NZFS populations.

Chapter 2 Earthquake impacts on a protected pinniped population in New Zealand

New Zealand fur seal pup at Otumatu, Kaikōura.

This chapter is a reformatted version of the following published manuscript:

Hall, A. A., Chilvers, B. L., Weir, J. S., & Boren, L. J. (2024). Earthquake impacts on a protected pinniped in New Zealand. Aquatic Conservation: Marine and Freshwater Ecosystems, 34(1), e4055.

MASSEY UNIVERSITY
TE KUNENGA KI PŪREHUROA
UNIVERSITY OF NEW ZEALAND

GRADUATE RESEARCH SCHOOL

STATEMENT OF CONTRIBUTION
DOCTORATE WITH PUBLICATIONS/MANUSCRIPTS

We, the student and the student's main supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the student's contribution as indicated below in the Statement of Originality.

Student name:	Alasdair Hall
Name and title of main supervisor:	Prof. Louise Chilvers
In which chapter is the manuscript/published work?	2

Describe the contribution that the student and members of the supervisory team have made to the manuscript/published work:[1]

All authors contributed to the study conception, methodology design, and data collection. Alasdair Hall and Louise Chilvers contributed to funding acquisition, data analysis, and visualization. The first draft of the manuscript was written by Alasdair Hall, and all authors commented on previous versions of the manuscript.

Please select one of the following three options:

- (●) **The manuscript/published work is published or in press**

  Please provide the full reference of the research output:

  Hall, A. A., Chilvers, B. L., Weir, J. S., & Boren, L. J. (2024). Earthquake impacts on a protected pinniped in New Zealand. Aquatic Conservation: Marine and Freshwater Ecosystems, 34(1), e4055.

- ( ) **The manuscript is currently under review for publication**

  Please provide the name of the journal:

- ( ) **It is intended that the manuscript will be published, but it has not yet been submitted to a journal**

Student's signature:	Alasdair Hall Digitally signed by Alasdair Hall Date: 2024.04.30 11:37:40 +12'00'	Main supervisor's signature:	Barbara Louise Chilvers Digitally signed by Barbara Louise Chilvers DN: cn=Barbara Louise Chilvers, c=NZ, o=Massey University, ou=Wildbase, School of Veterinary Science, email=b.l.chilvers@massey.ac.nz Date: 2024.10.15 08:26:50 +13'00'

This form should be placed at the beginning of each relevant thesis chapter.

[1] Refer to the Massey University Publishing and Authorship guidelines (OneMassey for staff, Stream for students) and/ or Contributor Roles Taxonomy (CRediT) guidelines for guidance.

2.1 Abstract

The impacts of natural disasters on marine mammals are poorly understood and difficult to study, which can hamper management responses following such events.

This study provides the first distribution and abundance assessment of New Zealand fur seal ('NZFS': *Arctocephalus forsteri*) colonies around Kaikōura, New Zealand, since a 7.8 magnitude earthquake in November 2016 caused substantial changes to both the local marine and terrestrial environments and led to the reconstruction of a major highway that runs adjacent to NZFS colonies.

Mark-recapture and direct counts in the 2022/23 breeding season estimated pup production for NZFS breeding colonies along the Kaikōura coast. Using established multipliers, pup estimates were used to provide the first comprehensive population estimate for Kaikōura's NZFS population since the earthquake.

Three new colonies and three new subcolonies were assessed and recorded, additional to reassessments of two established colonies. Overall, Kaikōura's NZFS population has grown and spread post-earthquake, with an upper total population estimate of between 21,560 – 28,327 animals in the 2022/23 breeding season. Some sites, such as Lynch's Reef, appear to have benefited from earthquake-induced coastal uplift, with pup production increasing. Contrastingly, the estimated 2,401 (± 99) pups produced at Ōhau Point in 2023 is similar to pre-earthquake estimates. This indicates that the earthquake has disrupted previously documented growth at this site. The distribution of NZFS breeding at Ōhau Point has also changed substantially since the last pre-earthquake assessment.

From our findings, alterations to the Ōhau Point New Zealand Fur Seal Sanctuary and similar protections at other locations on the Kaikōura coast are suggested, as greater numbers of

NZFS are now accessible to human interaction and disturbance. Our results demonstrate both how natural disasters and subsequent infrastructure modifications can impact coastal species, and how conservation measures may need to be amended accordingly.

2.2 Introduction

Understanding the size and distribution of wildlife populations is key to successful management (Robinson et al. 2011; Laidre et al. 2015). Without effectively monitoring such parameters, important population trends can be masked (McIntosh et al. 2018a), and interventions can come too late to achieve the desired outcomes (Voyles et al. 2014). Frequently, the logistics and costs of long-term monitoring programmes make them difficult to maintain (McIntosh et al. 2018a), and gaps in the record can be exposed following catastrophic events (Schwing et al. 2020; Southwell et al. 2022) as attempts are made to understand their ecological impacts.

Natural disasters can have a range of impacts on wildlife populations, the most immediate being mortality during the event itself. For example, an estimated 40,000 Hutton's shearwaters (*Puffinus huttoni*) were killed during the landslides caused by the Kaikōura earthquakes (Cuthbert 2019), while approximately 80% of Kangaroo Island's koala (*Phascolarctos cinereus*) population is thought to have perished in the 2019 – 2020 Australian bushfires (Dunstan et al. 2021). However, disaster-induced changes to habitats can also have long-term impacts on wildlife populations (Jolly et al. 2022). For example, nest counts in colonies of erect-crested penguins (*Eudyptes sclateri*) on the Antipodes islands in October 2014 showed significantly greater declines in the number of nests in breeding areas impacted by storms in January 2014 compared to colonies that were unaffected or only partially impacted (Chilvers and Hiscock 2019). The loss of habitat and resources likely also explains why bald eagles (*Haliaeetus leucocephalus*) and peregrine falcons (*Falco* *peregrinus*) failed to breed on an Alaskan island after a volcanic eruption (Williams et al. 2010). Following this 2008 eruption, Steller's sea lions (*Eumetopias jubatus*) were the only species to breed successfully one-year later (Williams et al. 2010), demonstrating the differential impacts of natural events on affected species.

The impacts of the 2016 Kaikōura earthquake on other local wildlife populations have been studied. Additional to the substantial mortality of Hutton's Shearwaters (Cuthbert 2019), there was mass mortality among benthic invertebrate species such as black-footed pāua (*Haliotis iris*) (Schiel et al. 2019) caused by the uplift and exposure of the benthos. The foraging distribution and dive behaviour of sperm whales (*Physeter macrocephalus*) in the Kaikōura Canyon also changed for at least a year post-earthquake, likely due to disturbances to marine food chains (Guerra et al. 2020). The longest-lasting impact of the 2016 earthquake to human populations was the damage to State Highway 1 (SH1), New Zealand's longest road. The earthquake damaged 194 km of SH1 (New Zealand Transport Authority 2017), and it took ca. 17 months for the road to re-open for travel at all hours in both directions, after the earthquake (Blake et al. 2019). Sections of the rebuilt road were moved away from the base of slopes, where landslips occurred, and closer to the coast (Green et al. 2018). At the geographical mid-point of the pre-earthquake Ōhau Point NZFS colony (Boren et al. 2006b), this shift was ca. 35 metres seaward.

New Zealand fur seals (*Arctocephalus forsteri*; 'NZFS') were hunted to the brink of extinction in the 18th and 19th centuries, with the population declining to less than 1% of the size it had attained prior to the arrival of humans in New Zealand (Emami-Khoyi et al. 2018). Since the cessation of sealing, NZFS have been expanding in area and numbers, recolonizing parts of their former range (Dix 1993; Bouma et al. 2008). However, colonies on the west coast of the South Island have experienced declining pup production since 1991 (Roberts and Neale 2016).

Estimating wildlife population abundances typically requires extrapolating data to produce an estimate for the wider population (Hammond 2001). In pinnipeds, pup production estimates are the most reliable indicator for the overall population size (Boren et al. 2006b; Roberts and Neale 2016; Chilvers 2021). Pups are the only instantly recognizable age cohort and are confined to the colony until they wean, meaning that this group can, theoretically, be counted in its entirety (Berkson and DeMaster 1985; Chilvers 2021).

With NZFS, abundance estimates involving pups have typically involved mark-recapture (Boren et al. 2006b; Chilvers 2021) and/or direct counts (Lalas and Harcourt 1995; Taylor et al. 1995; Gooday et al. 2018). While direct counts minimize disturbance to the animals, they may produce underestimates, as pups may be concealed from the surveyors. Therefore, this methodology can only provide insights into relative population sizes and trends, and only then if executed consistently (Watson et al. 2009). As such, mark-recapture provides the most accurate method for estimating pup production, and thus, population size (Boren et al. 2006b; Watson et al. 2009; Chilvers 2021). Mark-recapture involves capturing and marking a subset of pups at a site, and subsequently releasing them to mix with the remaining population (Boren et al. 2006b; Chilvers 2021). Following an interval of, typically, 24 hours, the colony is re-visited, and a walk-through conducted to count the number of marked and unmarked pups. This ratio can be used in the formula below to calculate pup production. For NZFS, marking is typically achieved through haircuts (Boren et al. 2006b), livestock markers (Chilvers 2021) or flipper tagging (Roberts and Neale 2016). The limited time between marking and recapturing means that mortality in the intervening period can be assumed to be zero (Chilvers 2021). However, dead pups observed during walk-throughs should be added to the live total to account for pre-survey mortality (Watson et al. 2009; Chilvers 2021).

Pups included in walk-through counts must be categorized with certainty as either marked or unmarked, or else discounted (Chilvers 2021). Similarly, marking and recapturing should be spread evenly through the study area, to help ensure that marked and unmarked pups have equal probabilities of being recaptured (Shaughnessy et al. 1995).

Despite its greater accuracy, mark-recapture is more expensive, labour intensive and disruptive to NZFS than direct counts (Watson et al. 2009). As such, calibration indices have been devised to estimate absolute pup production numbers from direct counts (Watson et al. 2009; Chilvers 2021). While estimates of pup production based on these indices are not as reliable as those calculated from mark-recapture, together with their respective confidence limits, they provide greater accuracy than untransformed direct counts (Watson et al. 2009).

Post commercial sealing, large aggregations of NZFS were first recorded in Kaikōura in the late 1950s (Street 1964), with breeding recorded from the 1990s at three sites: Ōhau Point, Lynch's Reef and Barney's Rock (Boren et al. 2006b) (Figure 2.1). The most recent local NZFS abundance study (Gooday 2016) estimated that 2,471 pups were born in the 2014/15 breeding season at Ōhau Point, an increase from the ~600 pups estimated in 2005 (Boren et al. 2006b). In their study, Boren et al. (2006b) determined that the Ōhau Point colony was growing at a rate of 32% per annum, while the Lynch's Reef colony, on the Kaikōura Peninsula, had a consistent annual pup production of 8 – 12 pups per year. In 2014, the Ōhau Point New Zealand Fur Seal Sanctuary was legally established under the Kaikōura (Te Tai o Marokura) Marine Management Act 2014. This remains New Zealand's only NZFS specific sanctuary and aims to protect local NZFS from anthropogenic disturbance by restricting access to viewing platforms. At the time of its establishment, the sanctuary covered about 4 hectares of land, however, this has changed following the 2016 earthquake and subsequent road reconstruction (see below). In addition to this specific spatial protection, all marine mammals in New Zealand are protected under the Marine Mammals Protection Act 1978, which makes it illegal to disturb, harass, injure or kill marine mammals.

No comprehensive assessments of Kaikōura's NZFS population have been conducted since a magnitude 7.8 earthquake struck the region in November 2016. This event had significant impacts on the terrestrial and marine habitats used by NZFS, including substantial uplift of coastal substrate by up to six metres (Alestra et al. 2019), and triggering over 10,000 landslides over ca. 10,000 km^2 (Massey et al. 2018). Additionally, 39×10^6 kg of benthic biomass was lost from the Kaikōura submarine canyon through an earthquake-induced flushing event (Mountjoy et al. 2018), disrupting local marine food chains (Guerra et al. 2020).

2.2.1 Study aims

Given the impacts of this earthquake and the associated reconstruction efforts on SH1, which runs adjacent to several NZFS colonies, an updated evaluation of Kaikōura's NZFS population abundance and distribution is required to assess whether current management initiatives continue to effectively protect the local population.

2.3 Materials and methods

Assessments of NZFS colonies were made along ca. 46 km of Kaikōura's coastline between Otumatu and Paparoa Point on the northeast of New Zealand's South Island (Figure 2.1). The definition of a NZFS colony described by Shaughnessy et al. (1994; 2015) is adopted here, whereby aggregations of breeding NZFS are referred to as colonies, and when such aggregations occur within two kilometres of each other they are categorized as a single colony. Distinct aggregations within colonies are referred to as subcolonies (Shaughnessy et al. 2015).

Under this definition, the study focuses on five breeding colonies, two of which consist of multiple subcolonies (Figures 2.2 – 2.4). The 'Ōhau Point' colony includes the pre-earthquake Ōhau Point colony (Boren et al. 2006b), hereafter 'Old Ōhau', which had covered a length of coastline of ca. 800 metres (Figure 2.2).

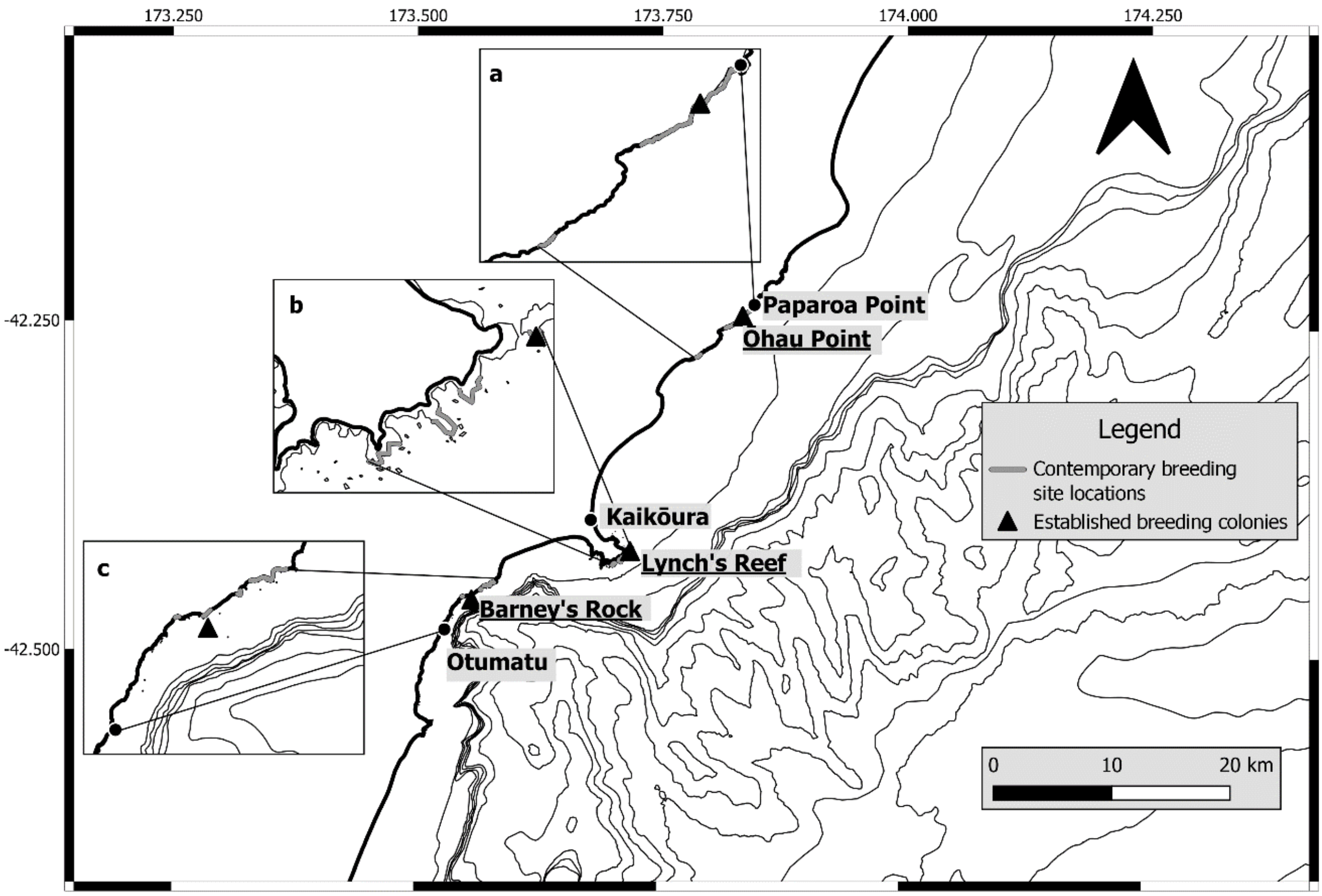


Figure 2.1 Map of the study area showing the location of the established breeding colonies: Ōhau Point, Lynch's Reef and Barney's Rock, and insets showing the locations of the breeding colonies a). north of Kaikōura, b). on the Kaikōura Peninsula and c). south of Kaikōura assessed in this study. The benthic topography off the Kaikōura coast is shown to indicate the position of the deepwater Kaikōura Canyon

2.3.1 Breeding colony distribution

The location of NZFS breeding within the study area was established using direct counts of pups in the 2021/22 and 2022/23 breeding seasons. Typically, these direct counts involved at least two surveyors walking past the colonies once and recording pup numbers using handheld tally counters or doing so from a point of elevation above the colony. When only one surveyor was available to survey, counts were conducted twice so a mean could be calculated.

The study area (Figure 2.1) was divided into three zones, ‘North of Kaikōura’ (between Paparoa Point and the northern end of Half Moon Bay; Figure 2.2) the ‘Kaikōura peninsula’ (between Kean Point and Whalers Bay; Figure 2.3) and ‘South of Kaikōura’ (between SC Subcolony 1 and Otumatu; Figure 2.4). These zones were assessed at least once per month during the NZFS breeding season (December, January and February) in both 2021/22 and 2022/23 to try to gain a full picture of where breeding takes place through time. Upon discovery of the South Rakautara colony in December 2021, this site was surveyed as frequently as possibly when surveying other sites north of Kaikōura. On the Kaikōura Peninsula, except for Whalers Bay, counts were conducted from the clifftops using binoculars. The offshore islet, Barney’s Rock, could not be accessed for counts. However, continued pupping at this site was confirmed through binocular scans from headland vantage points.

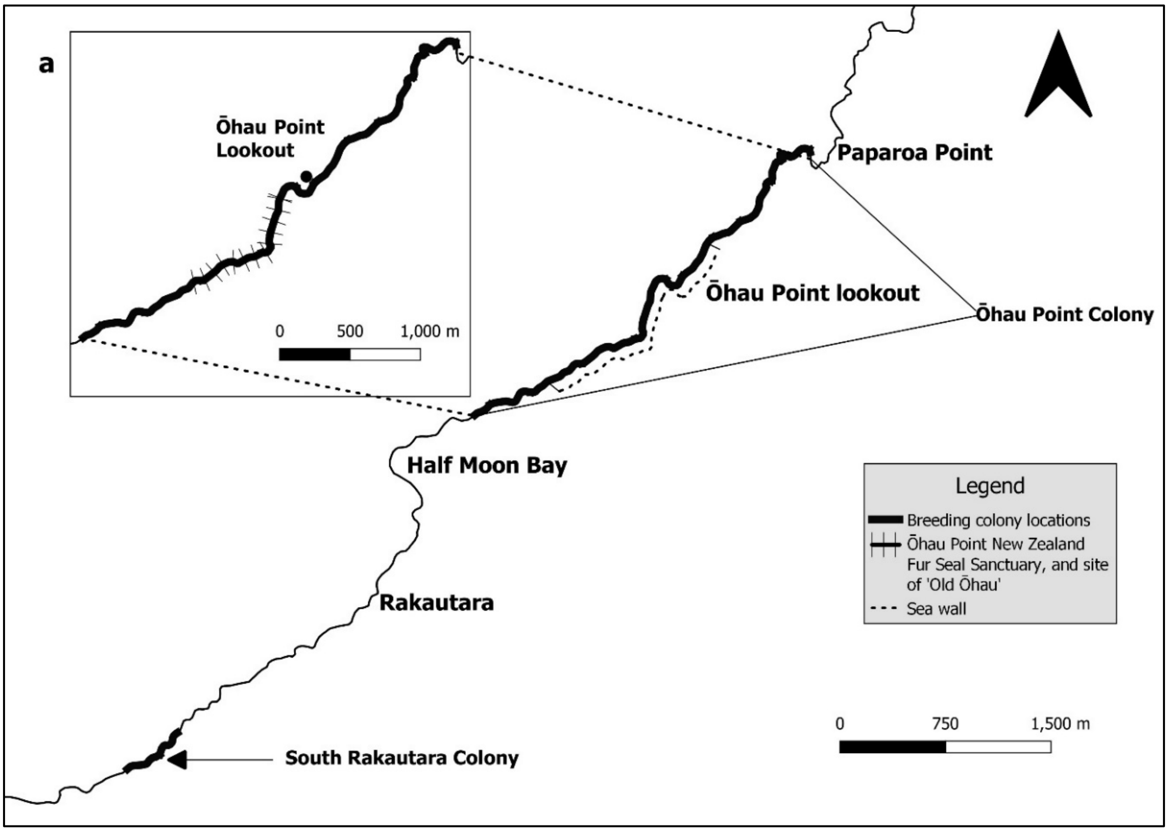


Figure 2.2 Location of Ōhau Point and South Rakautara NZFS colonies. The location of the sea wall is projected out from the coast for clarity

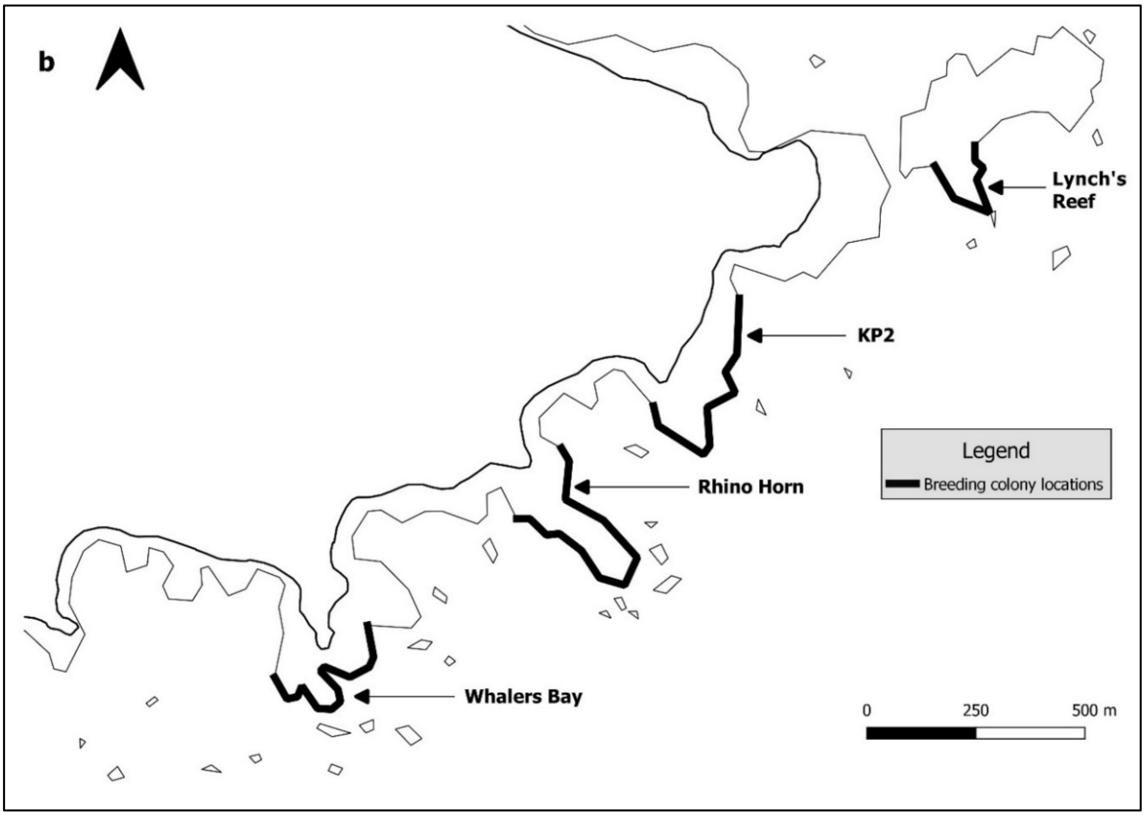


Figure 2.3 Location of Kaikōura Peninsula NZFS subcolonies

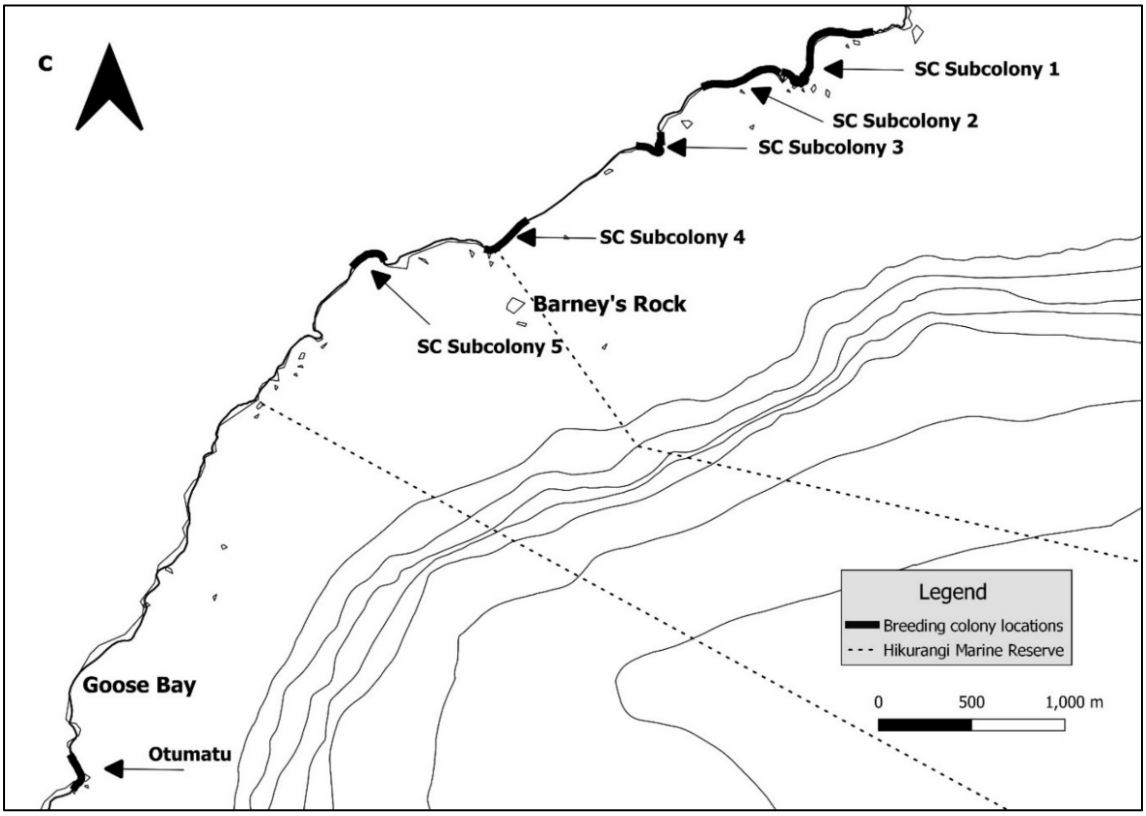


Figure 2.4 The location of the South Coast (SC) NZFS subcolonies and Otumatu breeding colony

2.3.2 Abundance estimates

Abundance estimates were calculated using direct counts and mark-recapture between February 16th and February 26th, 2023. Abundance estimates for NZFS typically take place from late January to March (Boren et al. 2006b; Chilvers 2021) as, at this stage, all pups will have been born and still have their instantly recognizable black pelage (Chilvers and Goldsworthy 2015).

These direct counts were additional to the previously described survey efforts and involved groups of 2 – 7 observers moving systematically through the colonies together and counting every NZFS pup they could find. This methodology was employed at the smaller South Coast subcolony sites (SC3, SC5) and Otumatu, where the small number of pups born (the largest single count was 37 ± 2 at SC3) made mark-recapture unnecessary, and at the Kaikōura Peninsula. While the pup numbers and terrain complexity of the Kaikōura Peninsula subcolonies warrant mark-recapture, the loss of fieldwork days due to Cyclone Gabrielle left insufficient time for mark-recapture here. Two sites, SC Subcolonies 1 and 4, were assessed using both direct counts and mark-recapture to enable the creation of calibration indices (Watson et al. 2009).

Mark-recapture analysis was conducted to obtain pup production estimates at larger sites. Pups were captured by hand, and marking was achieved through trimming a patch of fur on the pups' heads. Effort was made to spread marking effort evenly through each colony. All pups caught for marking were sexed to establish sex ratios. NZFS handling was subject to ethics approval by the Massey University Animal Ethics Committee.

Recaptures involved teams of four to five surveyors systematically moving through the colonies to count marked vs. unmarked pups (Boren et al. 2006b; Chilvers 2021). While this was done simultaneously, count independence was achieved by surveyors recording the numbers of marked and unmarked pups individually, using handheld counters. Surveyors avoided double counting pups that were running ahead of them by only counting animals as they were passed. Due to time constraints, only a single pass was made each site. Effort was made to check all potential hiding places for pups. The numbers of dead pups were counted and added to the final estimate of pup production (Gales and Fletcher 1999). The assumptions for the mark-recapture assessments were that (1) all pups had been born by February 17th (the first day of marking), (2) all pups at each site were available to be marked, (3) pups mixed well following marking, (4) marks were not lost before recaptures took place, (5) mortality could be assumed to be zero between marking and recapture, (6) no pups emigrated from or immigrated to each respective site during the period in which it being assessed (Gales and Fletcher 1999).

The numbers of marked and unmarked pups in each recapture sample were used to calculate a modified Petersen pup production estimate (Chapman 1952) as follows,

$$Pi = \left[\frac{(M+1)(Ci+1)}{Ri+1}\right] - 1$$

Here, for each replicate i, M represents the number of marked NZFS pups, C_i is the total number of pups included in the recapture sample and R_i is the number of marked pups recorded in the recapture sample. From this, the grand total estimate of pup production, P, is calculated from the mean of the Q Petersen estimators:

$$\frac{\sum_{i=1}^{Q} Pi}{Q}$$

Estimating pup production and extrapolating to a population estimate involved two calculations. First, mark-recapture estimates plus all direct counts undertaken at sites where mark-recapture was not done were summed to calculate a minimum pup estimate. Second, direct counts from sites where mark-recapture was not done were multiplied by 1.57 (the mean of the differences between the direct counts and mark-recaptures at SC1 and SC4), and this total was added to the sum of the mark-recapture estimates.

Standard errors for the mark-recapture estimates were calculated as (Gales and Fletcher 1999):

$$\sqrt{\frac{1}{Q(Q-1)\sum_{i=1}^{Q}(P_i - P)^2}}$$

The same formula was calculated for the standard errors of the direct count estimators, but in these calculations, Q represents the direct count estimates, rather than Petersen estimates.

Calculating population abundance estimates for NZFS has typically involved using multipliers to convert pup production estimates (Shaughnessy et al. 1994, 1995; Campbell et al. 2014; Chilvers 2021). The previously applied multipliers of 4.76 (Goldsworthy and Page 2007) and 4.9 (Taylor 1982) were used here.

2.4 Results

2.4.1 Breeding colony distribution

The distribution of NZFS breeding colonies in the 2022/23 breeding season along the Kaikōura coastline is shown in Figures 2.2 – 2.4.

Old Ōhau is now two-thirds narrower than it was pre-earthquake due to the relocation of SH1 closer to the coastline. The linear extent of the colony is now ca. 3.3 km, compared to the ca. 800 m length of Old Ōhau. This includes an extension of ca. 800 metres south of the southern boundary of Old Ōhau and ca. 1,770 metres north of the northern boundary of Old Ōhau.

The area available at Lynch's Reef has also increased from the ~500m^2 estimated by Boren et al. (2006b) to closer to ~10,000m^2, although not all of this is currently used for breeding.

South of Kaikōura, there are now two colonies, the South Coast colony and Otumatu. The South Coast colony includes six sub-colonies, the five new sites (SC1 – 5) shown in Figure 2.4, and Barney's Rock, which existed pre-earthquake. The mainland portion of the South Coast colony stretches along ca. 3.8 km of coastline, with gaps of less than two kilometres between each subcolony. Breeding at Otumatu occurs over ca. 180 metres.

2.4.2 Abundance estimates

The first new-born pups were recorded on December 8th in the 2021/22 breeding season, and December 9th in the 2022/23 breeding season (Appendix 1).

During mark-recapture, a total of 655 pups were marked across five sites (Table 2.1). The total estimated pup production for these sites was 3,609 ± 62 (Table 2.2). The total pup production at sites assessed through direct counts alone was 830 ± 13 (Table 2.2). Combined, the mark-recapture estimates and direct counts provided a minimum pup estimate of 4,400 – 4,478 pups (Table 2.2), with this range indicative of the mean ± the standard error.

Table 2.1 Numbers of NZFS pups marked at different sites along the Kaikōura coast

Site	No. of pups marked	No. of pups marked as percentage of total
Ōhau Point	384	15.9%
South Rakautara	60	21%
SC1	51	18.9%
SC2	110	25.7%
SC4	50	22%

When the direct count only sites were subjected to the 1.57 multiplier and added to the mark-recapture site estimates (Ōhau Point, South Rakautara, SC1, SC2 and SC4), the pup count estimate for Kaikōura was 5,703 – 5,781 (Table 2.2), with the range indicative of the mean ± the standard error. A regional population estimate of between 20,944 – 27,518 was calculated using Goldsworthy and Page's (2007) 4.76 multiplier, and a regional population estimate of between 21,560 – 28,327 was calculated using Taylor's (1982) 4.9 multiplier (Table 2.2).

Table 2.2 Kaikōura NZFS pup counts and mark recapture estimates in the 2022/23 breeding season

Location	Pup direct count mean ± SE	Pup mark-recapture estimate ± SE	Area specific pup production ranges
Ōhau Point		2,401 ± 99	
South Rakautara		285 ± 25	
Total North of Kaikōura		2,686 ± 97	2,589 – 2,783
Kaikōura Peninsula subcolonies:			
Lynch's Reef	41 ± 1		
KP2	159 ± 9		
Rhino Horn	323 ± 10		
Whalers Bay	245 ± 4		
Totals on Kaikōura Peninsula	768 ± 15		1,182 – 1,229
South Coast subcolonies:			
SC1	159 ± 1	269 ± 11	
SC2		427 ± 13	
SC3	37 ± 2		
SC4	157 ± 11	227 ± 4	
SC5	12		
Otumatu (separate colony)	13		
Totals South of Kaikōura	378 ± 7	923 ± 17	Minimum pup estimate[1] 977 – 993 Pup direct count to mark-recapture estimate (*1.57)[2] 1,074 – 1,090
Total	[3] 830 ± 13	3,609 ± 62	
Minimum pup estimate		4,400 – 4478	
Pup direct count to mark-recapture estimate (*1.57)[4]		5,703 – 5781	
Coarse estimated total population range (×4.76)[5]		20,944 – 27,518	
Coarse estimated total population range (×4.9)[6]		21,560 – 28,327	

Notes: Empty cells denote where no count was done for that type/location.

[1]Minimum pup estimate: equal to the sum of mark-recapture estimates plus all direct counts where mark-recapture was not undertaken.

[2]Direct count to mark-recapture estimate: equal to the sum of the mark-recapture estimates plus the sum of the direct counts where mark-recapture was not done multiplied by 1.57.

[3]This total represents the sum of the direct counts at sites where mark recapture was not also conducted.

[4] This total represents the sum of the mark-recapture estimates added to the sum of the products of each respective direct count multiplied by 1.57.

[5] This coarse estimated total population range equals the range between the minimum pup estimate multiplied by 4.76 and the direct count to mark-recapture pup estimate multiplied by 4.76 (Goldsworthy and Page 2007).

[6] This coarse estimated total population range equals the range between the minimum pup estimate multiplied by 4.9 and the direct count to mark-recapture pup estimate multiplied by 4.9 (Taylor 1982).

2.4.3 Sex ratios

Sex ratios between the colonies and subcolonies where marking took place are presented in Table 2.3. A chi-squared test was used to test for deviation from an expected 1:1 sex ratio between sites within the 2022/23 breeding season.

Table 2.3 NZFS pup sex ratios at the five sites where mark-recapture took place

Site	No. of pups sexed	Sex ratio (F:M)
Ōhau Point	384	1:1.19
South Rakautara	60	1:1
SC1	51	1:1.55
SC2	110	1:1.04
SC4	50	1:2.33

This test found no significant deviation ($\chi^2 = 6.6353$, df = 4, $P = 0.1565$).

2.5 Discussion

This study provides the first comprehensive population assessment of Kaikōura's NZFS since the November 2016 earthquake and subsequent road reconstruction.

2.5.1 Limitations of the study

Two multipliers were used to convert pup production estimates into population estimates. Taylor's (1982) multiplier derives from research from the Bounty Islands and is based on assumptions and knowledge of Antarctic fur seal (*Arctocephalus gazella)* vital rates (Payne 1977). Goldsworthy and Page's (2007) multiplier is derived from life-tables of South Australian NZFS. Evidently, applying these multipliers can only produce coarse population estimates with wide confidence intervals (Chilvers 2021). However, as many of the vital rate data used by Taylor (1982) and Goldsworthy and Page (2007) to calculate these multipliers do not exist for NZFS in New Zealand (Chilvers 2021), the more generalized multipliers have been adopted to make the results comparable with other studies.

The 1.57 multiplier was derived from direct counts and mark-recapture at SC1 and SC4. Ideally, a separate index would have been created to convert direct counts from the Kaikōura Peninsula, as the terrain here is different to that on the South Coast, which can impact index suitability (Watson et al. 2009). However, loss of study days due to Cyclone Gabrielle made it impossible to conduct mark recapture on the Kaikōura Peninsula. Future assessments of the Kaikōura Peninsula should attempt to create such an index as this would likely improve result accuracy.

2.5.2 Abundance estimates

Kaikōura's total NZFS abundance has increased since the last pre-earthquake assessment (Gooday 2016). However, pup production at Ōhau Point has not. There is some evidence for growth stalling at Ōhau Point prior to the 2016 earthquake, with Boren (*unpublished data*) estimating 2,390 (± 227) pups produced in 2011, comparable to the 2,471 estimated in 2015 (Gooday 2016). At that point, NZFS density within Old Ōhau was much greater than it is today across the larger post-earthquake breeding site, and Old Ōhau may have been approaching carrying capacity, with a concomitant reduction to the pup production growth rate (Roux 1987). With NZFS density at Ōhau Point now considerably lower than it was pre-earthquake and, given the overall NZFS population growth in Kaikōura and across New Zealand (Emami-Khoyi et al. 2018; Chilvers 2021), we would have expected Ōhau Point to revert to pup production growth. The fact that it has not likely relates to the extreme disruption to this colony caused by the 2016 earthquake. Much of the colony habitat was covered by landslip debris, causing loss of favourable habitat features (Gooday and Goldstien 2018), and likely mortality. Additionally, there was disturbance and NZFS mortality caused by the road reconstruction (NCTIR, *unpublished data*) with some NZFS relocated prior to the roadworks commencing, and the width of the colony reduced by the road's repositioning (Gooday and Goldstien 2018). Despite all of this, Ōhau Point has surpassed Gooday and Goldstien's (2018) expectations by achieving pre-earthquake pup production. Given the lower density of NZFS at Ōhau Point compared to before the earthquake, and the availability of suitable yet unused breeding habitat to the immediate north and south of the current breeding distribution, it is expected that pup production at the Ōhau Point colony could grow further. Lynch's Reef, the other site for which comparable pre-earthquake abundance data exist, produced substantially more pups in 2022/23 than it did pre-earthquake (Boren et al. 2006b).

The lack of recent population monitoring means it is impossible to establish the growth rate of the colonies studied here or estimate when previously unrecorded sites were founded. Gooday and Goldstien (2018) collated both published and unpublished NZFS data from the Kaikōura region going back to 1995. While these data show, for example, that NZFS were observed at Otumatu in 2001, no indication of whether they were adults or pups, or precise location data are provided. Direct counts in the 2021/22 and 2022/23 breeding seasons suggest that Kaikōura's pup production has remained relatively stable across these two years

(Appendix 1). However, data from other New Zealand NZFS colonies, and from New Zealand sea lion (*Phocarctos hookeri*) colonies on the Auckland Islands, indicate that 2022/23 was a low productivity breeding season (L. Boren, pers. comm.). While the causes of this are unknown, climate factors associated with a La Niña event may have contributed. That said, the productivity of the Kaikōura Canyon likely provides local NZFS colonies with some buffer against the changing environmental and biological conditions associated with El Niño-Southern Oscillation (ENSO) effects (Boren et al. 2006b), so it cannot be assumed that Kaikōura pup numbers were similarly impacted.

These uncertainties underscore the need for regular monitoring of wildlife populations. Gaps in the record are typical of NZFS monitoring across New Zealand (Chilvers 2021), and declining pup production at West Coast colonies (Roberts and Neale 2016) suggests that no single trajectory exists for New Zealand's NZFS colonies, demonstrating the need for more consistent and frequent assessments. It was fortunate, here, that Gooday's (2016) study provided relatively proximate 'before' data to allow inferences to be drawn as to the effects of the earthquake on Kaikōura's NZFS. As substantial natural disasters are uncommon and unpredictable, missing baseline data often impedes such analyses (Southwell et al. 2022). McIntosh et al. (2018a) describe the improvements made to population monitoring of Australian fur seals (*Arctocephalus pusillus doriferus*) when a coordinated, range-wide five-yearly census approach was adopted in 2002, replacing previous sporadic and opportunistic surveys. Currently, three-yearly NZFS mark-recapture monitoring is conducted at declining colonies on the West Coast of the South Island (Roberts and Neale 2016), and it is suggested that a similar programme is adopted in Kaikōura to continue assessments of the population's trajectory post-earthquake. Given the large interannual variation in *Arctocephalus* spp. pup production (McIntosh et al. 2018a), supplementary direct counts could also be conducted annually.

2.5.3 Breeding colony distribution

NZFS breeding distribution around Kaikōura has expanded beyond that described previously (Figure 2.1) (Boren et al. 2006b; Gooday 2016). The Ōhau Point colony is now ca. 3.3 km long, compared to the ca. 800m of Old Ōhau, although it is much narrower, and there is now a colony south of Rakautara, which had not been described previously. Similarly, the Kaikōura Peninsula currently supports three previously unrecorded subcolonies additional to Lynch's Reef (Boren et al. 2006b; Gooday 2016). South of Kaikōura, pre-earthquake, Barney's Rock was the only recorded breeding site within the study area (Boren et al. 2006b; Gooday 2016). Today, there is also breeding in the mainland South Coast colony, and at Otumatu.

There are several likely explanations for these expansions. First, New Zealand's NZFS population is continuing to recover from exploitation (Emami-Khoyi et al. 2018; Chilvers 2021). As such, the founding of new colonies is expected as resources are stretched at existing sites (Roux 1987). NZFS are highly philopatric (Bradshaw 1999; Chilvers 2021), and Bradshaw et al. (2000b) suggested that new colonies should develop near existing ones in a 'spillover effect', representing a compromise between philopatry and density-dependent factors that may impact breeding success. This has been demonstrated in NZFS colonies on Kangaroo Island, South Australia (Shaughnessy and Goldsworthy 2015), and newer subcolonies within the South Coast and Kaikōura Peninsula colonies also appear to follow this pattern. Breeding at SC4 is likely a spillover from Barney's Rock (Figure 2.4), an established breeding site with limited breeding space, directly offshore from SC4. SC4 had previously been an adult haul out site (Boren 2005), and thus appears to have followed a documented pattern whereby sites evolve from haul outs into breeding sites (Dix 1993; Ryan et al. 1997). Similarly, KP2, Rhino Horn and Whaler's Bay on the Kaikōura Peninsula may represent spill-over from Lynch’s Reef, another established and previously spatially limited breeding area (Figure 2.3). Alternatively, these sites could have been founded by migrants from elsewhere in New Zealand or from Ōhau Point post-earthquake.

Other sites, however, are unlikely to represent spillover. For example, South Rakautara is over five kilometres from the southernmost extent of Ōhau Point, and a substantial amount of apparently suitable, yet unused, breeding habitat exists between the two colonies. This is also true of the coastline between SC5 and Otumatu. An alternative explanation for these new colonies is immigration of NZFS, either from Ōhau Point, following the earthquake, or from elsewhere in the country (Robertson and Gemmell 2005). Immigrating animals from further afield would be uninfluenced by local philopatry when selecting breeding habitat (Bradshaw et al. 2000b).

Earthquake-induced changes to breeding habitats have also likely impacted NZFS breeding distribution. The earthquake caused variable uplift of up to 6 m at different locations along the coastline (Alestra et al. 2019; Hay 2020), at some sites creating additional breeding space. Pup production at Lynch’s Reef has increased from the consistent 8 – 12 pups between 2002 – 2005 (Boren et al. 2006b), which was believed to be close to the site’s spatial limit, to 41± 1 pups in the 2022/23 breeding season. However, uplift also has a potentially deleterious implication for NZFS at Lynch’s Reef. This subcolony is now substantially more accessible to people, as the uplift greatly reduced the depth of a channel that formerly separated the colony from a popular tourist carpark (Gooday and Goldstien 2018). As such, additional protection measures, perhaps in the form of sanctuary protections, should be considered to protect NZFS there from disturbance.

The earthquake, and subsequent anthropogenic responses, also had negative impacts on NZFS habitat quality at other sites around Kaikōura. For example, at Ōhau Point, uplift, combined with landslides dumping over 100,000 m^3 of debris (Barrett and Hayes 2019), destroyed important habitat features including caves. Additionally, SH1 was rebuilt closer to the ocean, decreasing the width of the habitat between the road and the ocean by two-thirds (Gooday and Goldstien 2018). The resultant loss of caves, crevices and rock pools led Gooday and Goldstien (2018) to downgrade the habitat's suitability, as per Taylor et al.'s (1995) scoring system, from nine (the highest possible) to six. The fact that the Ōhau Point colony is now ca. 4x the length of Old Ōhau (Figure 2.2), but that pup production today (2,401 ± 99) is comparable to the 2014/15 breeding season (2,471) (Gooday 2016), suggests that the breeding population is now distributed more linearly along the coastline, likely in response to the changes to the physical habitat. Additionally, NZFS were deliberately relocated prior to the commencement of roadworks on SH1 (Gooday and Goldstien 2018). As there is now insufficient room for the pre-earthquake breeding population within Old Ōhau, some relocated NZFS may not have returned.

Whether animals remain in a habitat impacted by a natural disaster depends on several factors. Immediately, members of the population must survive, although, even when populations are extirpated (Rodríguez-Lozano et al. 2015) they can later return and repopulate former habitats (do Rosário and Mathias 2007). Resource and habitat loss can also inhibit population persistence, even if members survived the initial event (Williams et al. 2010).

In Kaikōura, enough NZFS survived the earthquake, and enough local habitat remained suitable for the population to remain *in situ*. Indeed, NZFS bred at Ōhau Point immediately after the earthquake (Gooday and Goldstien 2018). How many NZFS died as a direct result of the earthquake is unknown, although its timing at around midnight on November 14th, 2016 (Furlong and Herman 2017), likely reduced mortality compared to if it had happened a few weeks later, or during daylight hours. Previous studies of Ōhau Point's colony dynamics indicate that few, if any, of that season's pups would have been born when the earthquake occurred (Boren 2005). Had the earthquake happened in late November or early December, there may have been substantial pup mortality, as the median pupping date at Ōhau Point was December 16th in in 2002, December 5th in 2003 and December 7th in 2004 (Boren 2005), and, once born, pups are initially relatively immobile (Crawley and Wilson 1976). Further, as NZFS are predominantly nocturnal foragers (Page et al. 2005), fewer adults would have been ashore when the earthquake occurred, relative to daylight hours. However, there would have almost certainly been mortalities in the initial earthquake-induced landslips. In particular, there may have been substantial numbers of heavily pregnant females ashore at Ōhau Point when the earthquake occurred (Boren 2005). Mortality amongst experienced female breeders may help to explain the continued stagnation of growth at Ōhau Point despite reduced population density. Discoveries of deceased individuals under rubble in 2017 suggests additional morality during subsequent landslips, and NZFS were also accidentally killed during the road reconstruction, for example, by heavy machinery (NCTIR, *unpublished data*).

While continued use of pre-earthquake sites suggests their continued suitability, despite the disruptions, philopatric species can make maladaptive decisions to remain in diminished habitats following natural disasters (Lai et al. 2007; Dudley et al. 2022). Given Ōhau Point's reduced habitat quality (Gooday and Goldstien 2018), this possibility should be considered for Kaikōura's NZFS. With amphibious marine species, the ongoing suitability of both marine and terrestrial habitats should be appraised following a natural disaster. The highly productive Kaikōura Canyon (De Leo et al. 2010), provides a year-round food source for species such as dusky dolphins (*Lagenorhynchus obscurus*) (Benoit-Bird et al. 2004) and sperm whales (*Physeter macrocephalus*) (Sagnol et al. 2015), and is within easy foraging range of NZFS from all colonies assessed in the present study (Boren 2005) (Figure 2.1). Its presence may explain why Kaikōura pups have previously been shown to be in better condition than those born in many other parts of New Zealand (Bradshaw et al. 2000a; Boren et al. 2006b), perhaps because females can make shorter foraging trips more energetically profitable, and thus nurse their pups more regularly and reduce pup fasting periods (Boren 2005). Pups at Ōhau Point also wean later than at other NZFS colonies in New Zealand and Australia (Boren 2005) despite being born at a similar time, potentially because females can remain in better condition while lactating, allowing them to extend the weaning period without severe energy penalties, and thus wean pups in better body condition (Boren 2005). While the earthquake-induced flushing of benthic biomass from the Kaikōura Canyon (Mountjoy et al. 2018) had observable impacts on local marine food chains, these effects appear to have been temporary (Guerra et al. 2020). Further, NZFS' foraging plasticity (Baylis et al. 2012) may have enabled them to adjust to any changes to their foraging habitat. The Kaikōura Canyon likely provides such a bountiful resource that, even if the quality of terrestrial habitats has been reduced, Kaikōura remains an excellent habitat for NZFS. The habitat suitability criteria (Taylor et al. 1995) that Gooday and Goldstien (2018) used to reassess Ōhau Point following the earthquake only considers terrestrial features, perhaps explaining why post-earthquake Ōhau Point has surpassed their expectations for pup production.

2.5.4 Implications for conservation

Changes to NZFS abundance and breeding distribution have implications for the management of the local population. Our study highlights the reality that human-NZFS interactions are likely to become more frequent and occur in different locations due to the numeric and spatial expansion of the local NZFS population.

The linear expansion of NZFS breeding at the Ōhau Point colony means that most breeding now occurs outside the boundaries of the Ōhau Point New Zealand Fur Seal Sanctuary

(Figure 2.2). This sanctuary was established under the Kaikōura (Te Tai ō Marokura) Marine Management Act 2014, with the intention of reducing anthropogenic impacts on the colony, in particular tourist disturbance. The entirety of Old Ōhau, which is contained within the sanctuary boundaries, is now far harder for tourists to access due to the construction of the 8–10 metre-high sea wall (Figure 2.2). However, post-earthquake breeding expansion means that substantial areas of the colony do not benefit from either the legal protections of the 2014 legislation, or the physical barrier of the sea wall, and are, therefore, at risk of disturbance. The Kaikōura (Te Tai ō Marokura) Marine Management Act 2014 is currently under a scheduled review, and it is recommended that the existing Ōhau Point New Zealand Fur Seal Sanctuary be extended.

Such extensions should not only incorporate where NZFS currently breed, but also consider where they may colonise next, should breeding continue expanding along the coastline. The narrowness of sections of the Ōhau Point colony, combined with the newly constructed sea wall, makes this site particularly vulnerable to sea level rise. The Old Ōhau section is the contemporary colony's narrowest point (Figure 2.2), and so it is likely that there will be continued redistribution of breeding to both the north and south of Old Ōhau if sea levels rise as predicted (Kulp and Strauss 2019). As such, to help futureproof the sanctuary, the entirety of Paparoa Point (Figure 2.2) should be considered for sanctuary protections. Currently, only the southern face of Paparoa Point experiences NZFS breeding, however, the entire Point features excellent NZFS breeding habitat (Ryan et al. 1997) and, given the typical spillover pattern of NZFS breeding colony expansion (Bradshaw et al. 2000b), this site is likely to be colonised in future. Similarly, at the southern end of the current Ōhau Point breeding distribution, we recommend that sanctuary protections are extended to the northern terminus of the Half Moon Bay seawall (Figure 2.2). The habitat available up to this point is very similar to that already occupied in the contemporary Ōhau Point colony and is also likely to be colonised in the future. Half Moon Bay itself is not, as it does not contain suitable NZFS breeding habitat.

Sanctuary protections would also be beneficial for some of the other Kaikōura NZFS breeding sites identified in this study that are at particular risk from human disturbance. In particular, NZFS on the Kaikōura peninsula are a draw for tourists (Acevedo-Gutiérrez et al. 2010) and these colonies are particularly accessible. People have also been observed walking into subcolonies SC1 and SC4 on the South Coast, where there is either no or minimal signage alerting people to the presence of NZFS, or existing regulations regarding interacting with these animals. If signage is employed to encourage compliance with current New Zealand regulations regarding maintaining appropriate distances from seals (20 metres), teleological signs have been shown to be more effective than ontological signs (Marschall et al. 2017).

Additional to providing human access to road-abutting colonies, SH1 is a direct threat to Kaikōura's NZFS (Hall et al. 2023 (Chapter 3)). The numbers of live and dead NZFS recorded on SH1 have increased from an annual average of 12 between 1996 – 2005 (Boren et al. 2008) to 59 (±16.9 s.d.) between 2012 – 2022 (Hall et al. 2023 (Chapter 3)). Much of this increase is likely due to local NZFS population growth, meaning the problem is likely to worsen without effective mitigation, such as more appropriate barriers, if local NZFS abundance continues to increase (Hall et al. 2023 (Chapter 3)). Continued increases to New Zealand's nationwide NZFS numbers (Emami-Khoyi et al. 2018; Chilvers 2021), and expansions of pinniped populations in other countries (Arnould et al. 2000; Shaughnessy and Goldsworthy 2015; Milano et al. 2020b) mean that mitigating similar deleterious interactions between such species and humans is likely to require greater consideration and resources in the future.

Chapter 3 Post-earthquake highway reconstruction: Impacts and mitigation opportunities for New Zealand pinniped population.

New Zealand fur seal mother and pup next to State Highway 1, south of Kaikōura.

This chapter is a reformatted version of the following published manuscript:

Hall, A. A., Chilvers, B. L., Weir, J. S., Vidulich, A., & Godfrey, A. J. R. (2023). Post-earthquake highway reconstruction: Impacts and mitigation opportunities for New Zealand pinniped population. Ocean & Coastal Management, 245, 106851.

MASSEY UNIVERSITY
TE KUNENGA KI PŪREHUROA
UNIVERSITY OF NEW ZEALAND

GRADUATE RESEARCH SCHOOL

STATEMENT OF CONTRIBUTION
DOCTORATE WITH PUBLICATIONS/MANUSCRIPTS

We, the student and the student's main supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the student's contribution as indicated below in the Statement of Originality.

Student name:	Alasdair Hall
Name and title of main supervisor:	Prof. Louise Chilvers
In which chapter is the manuscript/published work?	3

Describe the contribution that the student and members of the supervisory team have made to the manuscript/published work:[1]

Alasdair Hall, Louise Chilvers and Jody Weir contributed to the study conception and design. Material preparation, data collection and analysis were performed by Alasdair Hall, Ashley Vidulich and Jonathan Godfrey. The first draft of the manuscript was written by Alasdair Hall and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Please select one of the following three options:

- (●) **The manuscript/published work is published or in press**

  Please provide the full reference of the research output:

  Hall, A. A., Chilvers, B. L., Weir, J. S., Vidulich, A., & Godfrey, A. J. R. (2023). Post-earthquake highway reconstruction: Impacts and mitigation opportunities for New Zealand pinniped population. Ocean & Coastal Management, 245, 106851.

- ( ) **The manuscript is currently under review for publication**

  Please provide the name of the journal:

- ( ) **It is intended that the manuscript will be published, but it has not yet been submitted to a journal**

Student's signature:	Alasdair Hall — Digitally signed by Alasdair Hall Date: 2024.04.30 11:39:01 +12'00'	Main supervisor's signature:	Barbara Louise Chilvers — Digitally signed by Barbara Louise Chilvers DN: cn=Barbara Louise Chilvers, c=NZ, o=Massey University, ou=Wildbase, School of Veterinary Science, email=b.l.chilvers@massey.ac.nz Date: 2024.10.15 08:27:18 +13'00'

This form should be placed at the beginning of each relevant thesis chapter.

[1] Refer to the Massey University Publishing and Authorship guidelines (OneMassey for staff, Stream for students) and/ or Contributor Roles Taxonomy (CRediT) guidelines for guidance.

3.1 Abstract

Knowledge of how roads impact wildlife populations is limited but required to inform management and mitigation. Prioritising sites for mitigation involves identifying the most at-risk areas and populations, particularly after substantial changes to roading infrastructure.

We identify hotspots for New Zealand fur seal (NZFS: *Arctocephalus forsteri*) incidents (live or dead NZFS) on State Highway 1 (SH1) around Kaikōura, on New Zealand's South Island, and analyse whether cluster locations have persisted following earthquake-induced road reconstruction. We also assess spatial, environmental, and temporal influences on NZFS incidents.

Spatial records of incidents along SH1 were analysed to identify contemporary and former hotspots using Kernel Density Estimation Plus and a Poisson-based method. Spatial, temporal and environmental data were collected to assess these factors' effects on incident location and timing.

Between 2012 – 2022, an average of 59 incidents were recorded annually along 90 km of SH1. Ten significant hotspots accounted for 89% of incidents, along 2.75 km of road. Hotspot concentration shifted following road reconstruction. Incident numbers were significantly positively associated with traffic volumes and windspeed, and significantly negatively associated with temperature and rainfall. Road-abutting NZFS breeding areas explained most of the spatial variation in incidents.

SH1 is a threat to Kaikōura's NZFS, with its effects changing following an earthquake impacting NZFS distribution, and associated highway reconstruction. Hotspot analysis and current road protections suggest the risks could be substantially reduced by barrier construction along short stretches of road. This type of assessment should continue as climate

change raises sea levels and increases storm events globally. This analysis and mitigation approach could be used for any wildlife across numerous landscapes.

3.2 Introduction

Effectively managing wildlife populations involves understanding threats to their persistence at multiple spatial and temporal scales, from global risks such as climate change to localised stressors such as habitat loss (Boyd et al. 2008; O'Donnell et al. 2019).

3.2.1 Road ecology

Roads are one such stressor, and may impact wildlife in several ways, including mortality (Ramp and Ben-Ami 2006; Santos et al. 2018); reduced habitat quality (Huang et al. 2009; Koemle et al. 2018; Kunz et al. 2022) disrupted movement (Marsh et al. 2005; Northrup et al. 2012) and gene flow (Benson et al. 2016); and access provision to humans, which can facilitate deleterious interactions (Soofi et al. 2022).

Significant effort has been applied to understanding factors that contribute to wildlife-vehicle collision (WVC) hotspots (Bíl et al. 2019; Rendall et al. 2021; Özcan et al. 2022), and hot moments (Gonçalves et al. 2018; Schalk et al. 2019) – the latter referencing temporal and environmental factors associated with greater incidences of WVCs. Research has also assessed the efficacy of measures seeking to mitigate WVC occurrence (Denneboom et al. 2021; Kučas and Balčiauskas 2021).

Data for WVC studies come from sources including systematic surveys (Rendall et al. 2021; Wang et al. 2022), third party databases (Bíl et al. 2019), and call-out data for conservation bodies (Boren et al. 2008). Studies of small species typically require systematic surveys to ensure accuracy, but high detection rates for large mammals make non-systematic approaches (such as call-out data) comparable (Grilo et al. 2020). Regardless of methodology, the true proportion of a population involved in WVCs is likely to be underestimated, due to phenomena such as scavenging (Schwartz et al. 2018) and carcass decay (Hobday and Minstrell 2008).

There have been few studies of roads' impacts on pinnipeds (seals and sea lions), with exceptions including Boren et al.'s (2008) study on Kaikōura's New Zealand fur seal (*Arctocephalus* forsteri; NZFS), and a brief report describing vehicle collisions with northern elephant seals *(Mirounga angistirostris)* (Hatfield and Rathbun 1999). Several ways that roads can impact wildlife, such as impeding dispersion, are unlikely to be relevant to most pinnipeds, which typically remain near to the ocean while ashore. As such, this study focuses on road mortality, as this is likely to be SH1's most immediate impact on Kaikōura's NZFS.

NZFS lack natural terrestrial predators in New Zealand (Lalas and Harcourt 1995) but face threats when ashore including disease transmission (Boren 2005), and illegal killings (Department of Conservation 2022). The proximity of many of Kaikōura's NZFS to New Zealand's most significant road, State Highway 1 (SH1; Figure 3.1), represents an additional threat to be considered in spatial management decisions (Boren et al. 2008).

3.2.2 Kaikōura's NZFS and State Highway 1

Between 1996 – 2005, an average of 12 NZFS incidents were recorded annually on SH1, 40% of which were roadkill mortalities (Boren et al. 2008). Pups accounted for 25% of NZFS incidents, and 61% of fatalities, however these figures increased through time.

New Zealand's NZFS population is growing as the species recolonises areas from which they were extirpated by hunting in the 18th and 19th centuries (Lalas and Bradshaw 2001). Since Boren et al. (2008), Kaikōura's NZFS colonies have increased from an estimated ~600 pups produced at Ōhau Point in 2005 (Boren et al. 2006b) to 2,471 in 2015 (Gooday 2016). In addition, NZFS now breed both to the north and south of the locations described by Boren et al. (2008) (Hall et al. 2024 (Chapter 2)) (Figure 3.1).

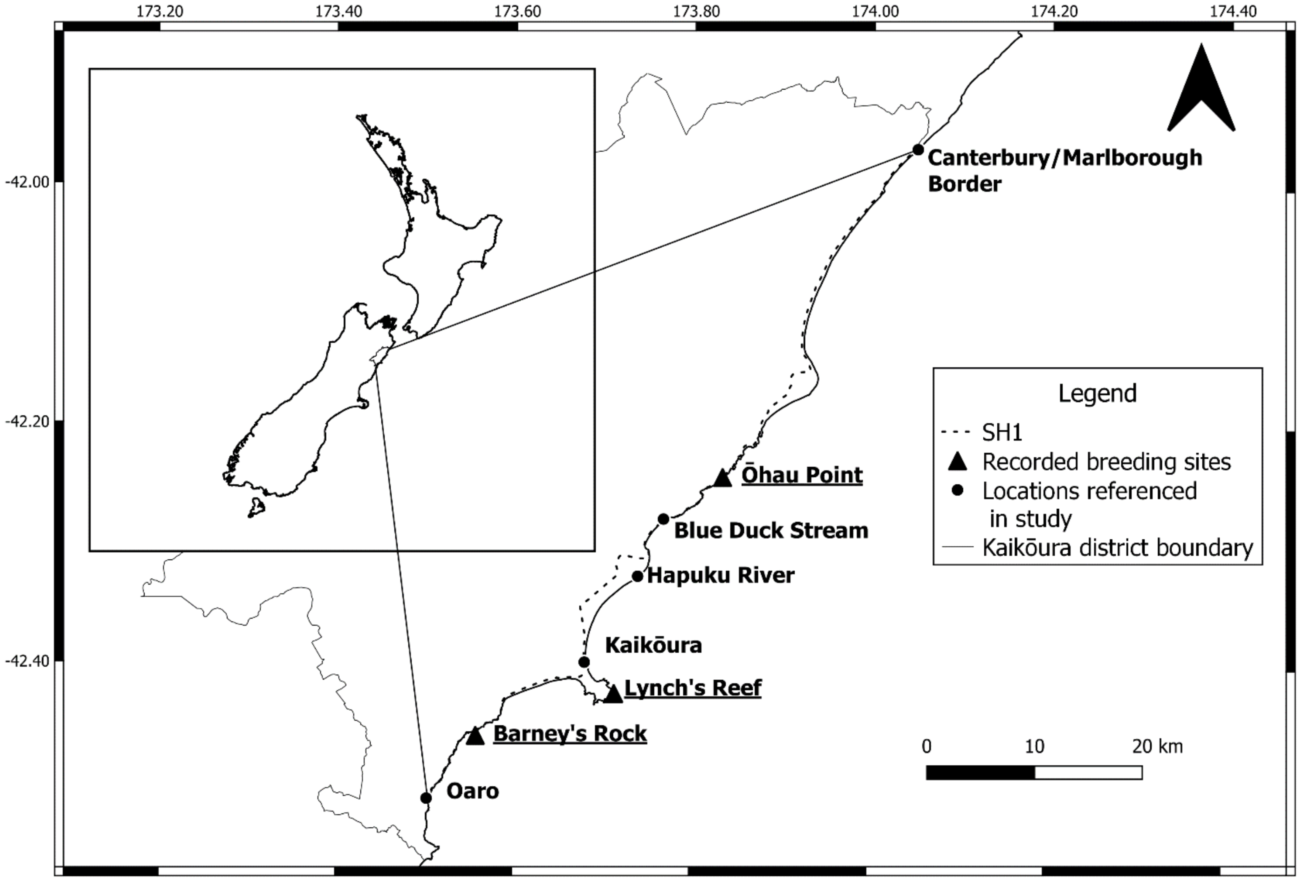


Figure 3.1 Key locations in the study area, including the three previously recorded New Zealand fur seal breeding sites in the Kaikōura region: Ōhau Point, Lynch's Reef and Barney's Rock (Boren et al. 2008)

SH1's physical infrastructure has also changed substantially since 2005, largely due to reconstruction following a Mw 7.8 earthquake that struck the area in November 2016 (Aghababaei et al. 2020). The road was relocated away from the base of inland slopes and closer to the coast, atop newly created seawalls, and roadside barriers were altered from those described by Boren et al. (2008) (discussed below).

3.2.3 Study aims

We aimed to assess the current effects of SH1 on Kaikōura's NZFS, and to inform possible mitigation measures. This knowledge is not only important for the health of the NZFS population, but also for discerning the dangers posed to motorists.

We first examine the recent trends in NZFS road incidents on SH1, and subsequently undertake analysis to meet the following aims:

1. Where are the contemporary hotspots for NZFS accessing SH1?
2. Which factors contribute to the spatial variability in where NZFS currently access SH1?
3. Have hotspot locations changed over time?
4. Which factors contribute to the temporal variability in when NZFS access SH1?

3.3 Materials and methods

3.3.1 Study site

This study focused on ~90 km of SH1 between Oaro and the north-eastern extent of the Canterbury region, where it borders the Marlborough region on the east coast of New Zealand's South Island (Figure 3.1). SH1 is New Zealand's longest and most significant national highway, running the length of the country. In the Kaikōura region, it is a two-lane, bidirectional highway which, in many places, runs within metres of the coastline. The relevant portion of SH1 experiences an annual average daily traffic (AADT) volume of greater than 2,500 vehicles, of which approximately 10% are heavy vehicles (gross vehicle mass of over 3.5 tonnes) (Aghababaei et al. 2020). The methodology is described in sections relating to the research aims in Section 3.2.3.

3.3.2 NZFS road incident data collection

Data on live or dead NZFS on SH1 (‘NZFS incidents’), were collected by the New Zealand Transport Authority (NZTA). Since 2012, NZTA has been permitted by the New Zealand Department of Conservation (DOC) to both herd live NZFS off SH1 and dispose of NZFS carcasses on SH1. This is typically done in response to callouts, or while road maintenance is being undertaken. As such, it is unlikely that any NZFS incident reflects the time, and perhaps also the location, where the animal first entered the road. NZTA staff record each incident and produce annual reports. This study analyses data from May 2012 – April 2022. There are, however, recording gaps, most notably between the November 2016 earthquake until June 2018, when data were collected by another body in a different format.

To spatially record NZFS incidents, NZTA uses a linear referencing system (Location Referencing Management System; ‘LRMS’). When a NZFS incident occurs, the location is recorded using Mobile Road (current version 3.2.3, NZTA, Wellington) a smart device app.

3.3.3 Contemporary NZFS incident hotspot identification

Bíl et al.’s (2013, 2016, 2019) Kernel Density Estimation Plus (KDE+) method was used to identify NZFS incident hotspots. We used the KDE+ software package (vers. 2.5, Transport Research Centre, Olomouc, Czechia) available for download as freeware (http://www.kdeplus.cz/en/download), which analyses between-intersection segments of road (Bíl et al. 2016; 2019). Two input CSV files were created; one containing the lengths of the between-intersection road segments, and the second the locations of NZFS incidents in each segment, given by their distance from the segment start point.

For the contemporary hotspot analysis, NZFS incident data were limited to January 2021 – April 2022 to reflect the current state of SH1’s infrastructure. Measurements of between intersection segment lengths and the position of incidents were conducted in QGIS (QGIS Development Team 2022).

In the KDE+ software, a kernel bandwidth of 150 units was chosen, and ‘Accurate (GPS)’ was selected under the software’s ‘Data Accuracy’ options.

3.3.4 Current spatial drivers of NZFS incidents on SH1

SH1 was divided into 359 250-metre segments (Rendall et al. 2021) to determine factors that may explain the spatial distribution of NZFS incidents. For each segment, a binary response variable was created indicating whether it contained at least one NZFS incident. Again, the NZFS incident data were limited to January 2021 – April 2022.

Twenty-six road segments contained at least one NZFS incident, and, to permit modelling of spatial variables, 52 incident-free segments were randomly selected for comparison. To qualify for selection, an incident-free segment had to run within 100 metres of a beach head (measured in QGIS) in at least one location, and not be in a residential area, to reduce any likelihood of selecting segments with multiple obstructions between the road and the coast. For each segment, a suite of variables was recorded for use in the models (Tables 3.1 & 3.2).

Table 3.1 Continuous variables analysed in the assessments of the spatial drivers of New Zealand fur seal incidents

Variable	Values
Road width	Metres
Distance to the nearest bend	Metres
Distance to the nearest stream/culvert	Metres
Road segment height	Metres
Distance to beach head	Metres

Table 3.2 Categorical variables analysed in the assessments of the spatial drivers of New Zealand fur seal incidents

Variable	Values
Roadside vegetation coverage (within 5 metres of road)	0-24%, 25-49%, 50-74%, 75-100%
Location of railway line relative to road	Coastal/inland
Road segment abuts known NZFS breeding area	Yes/no
Coastal substrate type	Sand or pebble beach/rocky shore
Non-barriered road	Present/absent
Single guard rail	Present/absent
Single guard rail combined with plastic mesh safety fencing (PMSF)	Present/absent
Motorcycle protection railing (MPR)	Present/absent
Agricultural fencing	Present/absent
Standalone PMSF	Present/absent
Sea wall	Present/absent

Roadrunner (Argonaut Ltd, Wellington) was used to determine the presence/absence of different road barrier types and sea walls, the percentage coverage of roadside vegetation, the relative position of the railway tracks and the distance to the nearest bend capable of visually obstructing a driver. Roadrunner is a dash-cam style video viewer used by NZTA, with an associated indication of location, based on the LRMS. Segment heights were recorded using BackCountry Navigator PRO v. 7.2.8, (CritterMap Software LLC, Washington), and were measured by AAH at segment mid-points. Road width measurements were taken in Mobile Road v. 3.2.3 (NZTA, Wellington), also at segment mid-points. QGIS was used to measure distance to the nearest culvert/stream and beach head from segment midpoints, while distance to the nearest bend was measured in RoadRunner, as this was deemed to provide a more accurate method of identifying the bend closest to a section midpoint which would obscure a driver's vision. All measurements were taken from section midpoints. Coastal substrates were determined using Google Earth®. The presence/absence of NZFS breeding sites was based on direct counts of pups in the 2021/22 breeding season (Chapter 2). Ideally, slope would have been included as a variable in this analysis, however, collecting this data was deemed unsafe for researchers due to the risk posed by traffic, and so segment height was used instead (Appendix 2).

Pearson correlation coefficients tested for correlations among continuous explanatory variables and the response variable. Initially, this included continuous measurements of the lengths of different barrier types, which were later converted to categorical binary variables (Table 3.2).

Chi-squared tests were used to determine any statistically significant relationships between categorical variables.

Low to moderate associations existed between many of the explanatory variables and the response. These relationships were explored further using chi-squared tests and cross tabulated data between the categorical explanatory variables and the response. The test results indicated that one explanatory variable explained such a large majority of the spatial variation in NZFS incidents that a full logistic regression was not required. All statistical testing was conducted in R (R Core Team 2022).

3.3.5 Hotspot persistence analysis

To assess 2012 – 2022 hotspot persistence, incidents were aggregated into kilometre segments (Lee et al. 2006) and their observed spatial distribution compared with the expectation from a random scenario, where the probability of incidents per segment should follow a Poisson distribution (Boots and Getis 1988).

Kilometre segments were used as, prior to 2019, this was the typical precision with which incidents were recorded. As the analysis requires equal-length segments (Malo et al. 2004), some stretches of SH1 were excluded where they did not constitute complete kilometres. Only one NZFS incident was omitted as a result.

This methodology (Malo et al. 2004; Lee et al. 2006) requires calculating the mean number of incidents per segment (λ), allowing the probability of a segment containing a given number of incidents (x) to be calculated as follows:

$$p(x) = \frac{\lambda^x}{(x!\,e^\lambda)}$$

Due to the high number of zero incident segments, the threshold for significance was low with an α of 0.05, so an α of 0.01 was selected (Lee et al. 2006). Separate analyses were conducted for each period meaning that, while 2012, 2016 and 2022 were not complete calendar years, each segment always had an equal opportunity to accumulate incidents. Data for 2017 were not available, and 2018 was excluded as only four incidents were recorded. Data were also omitted where incident locations were illegible, or where precision was insufficient.

Despite the α of 0.01, there was still a low threshold for significance, so the analysis was repeated including only segments containing at least one incident.

3.3.6 Temporal and environmental drivers of NZFS incidents on SH1

To model environmental and temporal drivers of NZFS incidents, daily values of traffic volume, wind speed (km/hour), precipitation (mm), average temperature (°C), tide height (m), and significant wave height (m) were obtained and averaged across the month from which they were recorded for use as model covariates. An additional season variable was created to determine whether season impacted NZFS incident numbers. All available NZFS incident records made since data collection restarted post-earthquake in June 2018 were analysed.

Traffic volume was extracted from NZTA's Traffic Monitoring System and originated from a telemetry site north of the Hapuku River (ID: 01S00144; Figure 3.1). Wind speed, precipitation and temperature data were extracted from the National Institute of Water and Atmospheric Research (NIWA)'s CliFlo database (https://cliflo.niwa.co.nz/) and originated from a weather station on the Kaikōura Peninsula (no. 4506, name: Kaikōura Aws). Maximum daily tide height (metres above lowest astronomical tide) for each 24-hour period in Kaikōura was extracted from NIWA's Tide Forecaster (https://niwa.co.nz/information-services/tide-forecaster). Significant wave height was extracted from the Historic Forecast of MAGICSEAWEED.com (https://magicseaweed.com/). A daily average significant wave height (the largest third of all waves) was derived from the mean of the eight three-hourly forecasts per day. Forecasts were taken from the Blue Duck Stream site (Figure 3.1).

Poisson regression models were used to identify environmental and temporal drivers of NZFS incidents on SH1.

Pearson correlation coefficients tested for correlations among continuous explanatory variables and the response variable, and ANOVAs investigated possible correlations between the season variable and the continuous variables. Variance inflation factor (VIF) tests were used to test for multicollinearity.

Due to the time series nature of the data, models with and without the season variable were compared to determine the impact on autocorrelation. Autocorrelation function plots and Durbin-Watson tests were used to determine whether autocorrelation was a concern in each model.

The substantial difference between the mean (4.4) and the variance (60.6) of the response variable suggests overdispersion. To determine whether this was problematic, a comparison of residual deviance and degrees of freedom, and a dispersion test on the fitted model were conducted. A residual deviance and degrees of freedom with a ratio around 1:1 is considered appropriately dispersed. If the ratio is larger, and the dispersion tests yield a significant p-value, a Quasi-Poisson model could be considered an appropriate alternative. Quasi-Poisson models were compared to the standard Poisson model to determine whether there was a considerable difference, however final model selection was based on the dispersion test.

Diagnostic plots assessed whether there was a linear relationship between the log value of the response and the explanatory variables.

3.4 Results

3.4.1 Overview

Between May 2012 – April 2022, 393 dead and 57 live NZFS were recorded on SH1 within the study area in 307 NZTA records (Figure 3.1; Table 3.3).

For years with data for all 12 months ('complete years'; 2013 – 2015 incl. and 2019 – 2021 incl.), an annual average of 49 (±9.2 s.d.) dead and 9 (±9.8 s.d.) live NZFS were recorded. The average number of NZFS incidents in complete years was 59 (±16.9 s.d.). The most NZFS were killed in 2020 (59) while the fewest NZFS (37) were killed in 2019. The number of NZFS killed on SH1 between January – April 2022 (36) was higher than all the previous years for which comparable data are available (n=7).

Table 3.3 Interannual comparison of New Zealand fur seal (NZFS) incidents on State Highway 1, Kaikōura, New Zealand. * denotes complete years' worth of data

Year	No. of live NZFS on SH1	No. of dead NZFS on SH1	Total NZFS incidents	Pups as percentage of annual road mortality
2012	2	21	23	60%
2013*	3	40	43	97.1%
2014*	17	52	69	94.1%
2015*	25	58	83	58.6%
2016	0	34	34	84.4%
2018	1	3	4	100%
2019*	1	37	38	94.3%
2020*	6	59	65	96.6%
2021*	2	53	55	96.2%
2022	0	36	36	97.2%

For incidents with an associated indication of the individual's age cohort (n=387), pups accounted for 83.7% of NZFS recorded on SH1. For mortalities with an associated indication of the individual's age cohort (n=350), pups accounted for 87.8% of NZFS road mortality on SH1. In the final three complete years, pups accounted for an average of 95.7% of NZFS road mortality.

3.4.2 Contemporary NZFS incident hotspot identification

Forty-seven between-intersection segments of SH1 (labelled A through UU) were analysed, totalling 88.24 km. From the 91 NZFS incidents between January 2021 – April 2022 (incl.), 10 significant clusters were identified, accounting for 89% of the analysed incidents (81/91).

Significant cluster sizes ranged from two incidents (n=2, O1 and O3) to 35 (n=1, TT1) (Figure 3.2; Table 3.4). Cluster lengths ranged from 90 metres (O3) to 1.74 kilometres (TT1) (Table 3.4). The combined cluster length was 2.75 kilometres, 3.1% of the analysed portion of SH1. The strongest cluster was TT1 (cluster strength = 0.671, incident count = 35), and the weakest was TT5 (cluster strength = 0.108, incident count = 4) (Table 3.4).

The KDE+ software provides a more stringent test called the segment test (Bíl et al. 2016). Four clusters met the threshold for the segment test: O2, O4, Q1 and TT1 (Figure 3.2; Table 3.4).

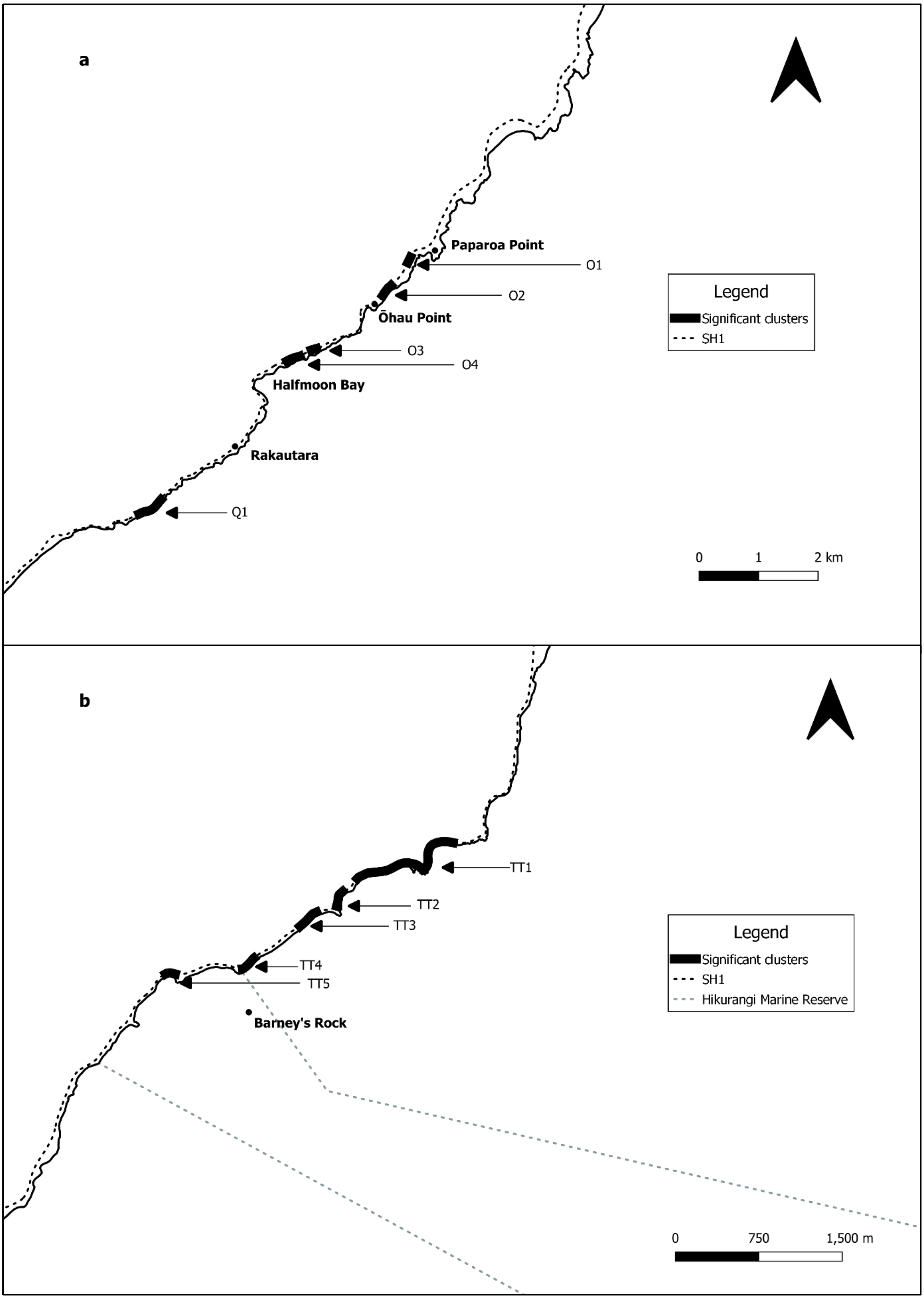


Figure 3.2 Locations of significant clusters of New Zealand fur seal road incidents a). north of Kaikōura and b). south of Kaikōura, New Zealand

Table 3.4 Statistically significant clusters of New Zealand fur seal incidents identified using Kernel Density Estimation Plus

Road section ID	Cluster ID	No. incidents/cluster	Cluster strength	Cluster length (metres)	Incident density per 100 metres of cluster	Segment strength
O	O1	2	0.125	100	2	0
O	O2	4	0.559	220	1.8	1
O	O3	2	0.125	90	2	0
O	O4	5	0.601	230	2.2	1
Q	Q1	11	0.671	460.62	2.4	1
TT	TT1	35	0.673	1073.81	3.3	1
TT	TT2	6	0.348	150	4	0
TT	TT3	7	0.434	193.73	3.6	0
TT	TT4	5	0.133	145.89	3.4	0
TT	TT5	4	0.108	90	4.444	0

3.4.3 Spatial drivers of NZFS incidents on SH1

Pearson correlation tests found several predictor variables showing low-moderate, statistically significant correlations. These included: distance to beach head and distance from nearest bend (Pearson correlation coefficient = 0.69, $p < 0.001$), distance to beach head and road width (Pearson correlation = -0.52, $p < 0.001$) road width and distance to nearest bend (Pearson correlation coefficient = - 0.4, $p < 0.001$), single guard rail and non-barriered road (Pearson correlation coefficient = - 0.763, $p < 0.001$), MPR and non-barriered road (Pearson correlation coefficient = - 0.388, $p < 0.001$), sea wall and non-barriered road (Pearson correlation coefficient = - 0.309, $p < 0.01$), sea wall and single guard rail (Pearson correlation coefficient = - 0.333, $p < 0.01$) and sea wall and agricultural fencing (Pearson correlation coefficient = 0.239, $p < 0.05$).

Chi-squared tests revealed statistically significant relationships between several categorical variables, including: known breeding area and single guard rail combined with PMSF (binary) ($x^2 = 5.567$, p = 0.018), known breeding area and MPR (binary) ($x^2 = 9.152$, p=0.002), MPR railing and agricultural fencing ($x^2 = 4.537$, p = 0.033), agricultural fencing and sea wall ($x^2 = 10.954$, p = 0.001)

Additional chi-square tests of the binary variables and the response, and the cross-tabulated data indicated that known breeding area explained most of the variation in NZFS incidents. Cross-tabulation indicated that known breeding area correctly classified NZFS incidents in 74 out of 78 observations, and the chi-squared test between NZFS incidents and known breeding area was statistically significant ($x^2 = 57.123$, $p<0.00001$). The significance of this chi-squared result is sufficient to represent the relationship between the two variables.

3.4.4 Hotspot persistence analysis

Both persistence analyses used data from 2012 – 2022, with 428 NZFS incidents available for analysis. Across the study period, 25 segments contained at least one NZFS incident.

The first persistence analysis returned 39 non-unique significant segments. The most persistent individual kilometre segment, 118.12, was significant in the eight analysed years between 2012 – 2021 (incl.), and contained the greatest number of incidents (78, 18.2% of the total), followed by 118.11, 163.06, 163.07 and 163.08 which were all significant in five of the analysed years (Table 3.5; Figure 3.3). Of these, 163.06 contained the greatest number of incidents (64, 14.9% of the total). Of the significant segments from the period prior to the 2016 earthquake (May 2012 – November 2016 incl.), 70% (14) were north of Kaikōura. By contrast, following the earthquake (January 2019 – April 2022), 63.2% (12) were south of Kaikōura.

Table 3.5 New Zealand fur seal incidents recorded in different kilometre sections of State Highway 1 through time. * Denotes values that were significant in the respective year in the analysis involving all 86 sections, ** denotes values that were significant in the respective year in the second persistence analysis (involving only the 25 sections with at least one incident)

Section	Year									
	2012	2013	2014	2015	2016	2018	2019	2020	2021	2022
90.07	0	0	0	0	0	0	0	0	1	0
118.06	0	0	0	1	0	0	0	0	0	0
118.07	0	0	0	1	0	0	0	0	0	0
118.08	0	0	0	0	1	0	0	0	0	0
118.09	2	0	0	0	0	0	0	0	0	0
118.10	1	0	0	0	0	0	0	0	0	0
118.11	5**	1	2	14**	11**	0	4*	10**	1	2
118.12	6**	7**	15**	14**	15**	0	4*	11**	4*	2
118.13	2	10**	19**	11**	1	0	1	0	2	0
118.14	2	10**	13**	7*	0	1	0	0	5*	0
118.15	1	0	1	1	0	0	0	0	0	0
118.16	0	1	1	0	0	0	0	0	0	0
118.17	0	0	0	1	0	0	0	1	0	2
118.18	0	1	1	3	2	0	0	1	3	7**
118.19	0	0	0	2	0	0	0	0	0	0
163.05	0	0	0	2	1	1	0	3	0	0
163.06	0	1	0	4*	0	0	10**	17**	18**	14**
163.07	0	3	3	4*	0	0	7**	15**	12**	4*
163.08	2	6**	6*	4*	0	1	9**	3	3	4*
163.09	0	0	0	0	0	0	2	4*	4*	1
163.10	0	0	0	0	0	0	0	0	1	0
163.11	0	0	0	4*	0	0	0	0	1	0
163.12	1	2	1	0	0	1	0	0	0	0
163.13	0	1	0	0	0	0	0	0	0	0
163.15	1	0	2	1	0	0	0	0	0	0

The second persistence assessment returned 25 non-unique significant segments. Segment 118.12 (Figure 3.3a) was the most persistently significant, meeting or surpassing the relevant threshold in six years (2012 – 2016 (incl.) and 2020) (Table 3.5). This was followed by 118.11 and 163.02, which were both significant in four of the analysed years, and then by 118.13 and 163.07, which were both significant in three of the analysed years (Table 3.5) Of the 14 non-unique significant segments prior to the 2016 earthquake, 92.9% (13) were north of Kaikōura. Of the 11 non-unique significant segments post-earthquake, 72.7% (8) were south of Kaikōura.

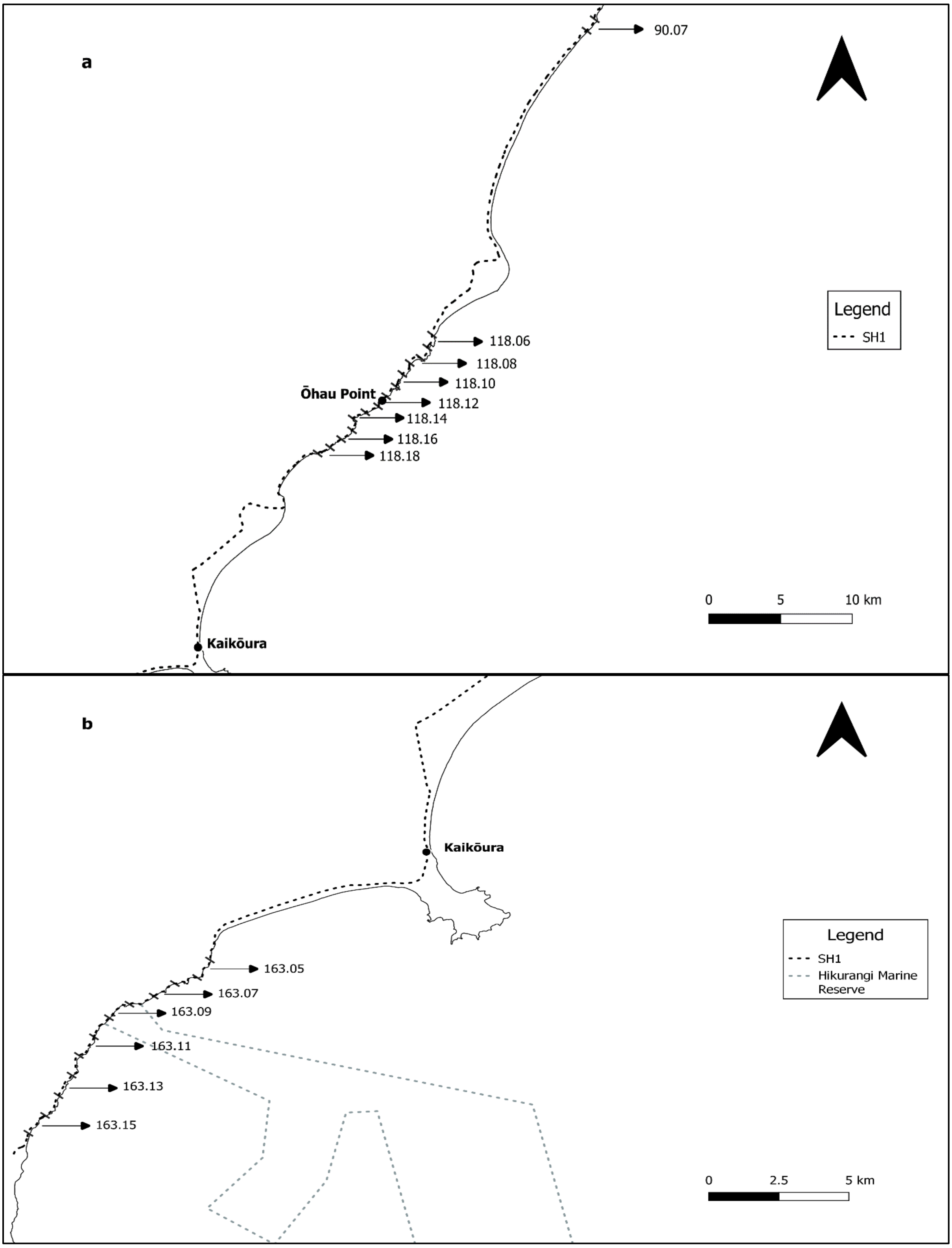


Figure 3.3 Kilometre segments containing at least one New Zealand fur seal incident, a). north of Kaikōura and b). south of Kaikōura, New Zealand. For contiguous sections, every other section is labelled

3.4.5 Temporal and environmental drivers of NZFS incidents on SH1

Pearson correlation tests found a moderate correlation between average traffic and average daily temperature (Pearson correlation coefficient = 0.655, $p < 0.001$), and a weak significant correlation between wind speed and rainfall (Pearson correlation coefficient = 0.431, $p < 0.01$). ANOVAs suggested temperature ($F = 58.540$, $p < 0.001$) and traffic volume ($F = 9.024$, $p < 0.001$) were significantly correlated with season.

Durbin-Watson tests for autocorrelation suggested that the model without the season variable showed evidence of autocorrelation. However, the addition of the season variable addressed this concern, with the autocorrelation plot and the Durbin-Watson test outputs within the desired thresholds.

None of the variables subjected to VIF tests showed concerning multicollinearity (VIF greater than 5; (Alauddin and Nghiemb 2010)). While the overdispersion test showed that the residual deviance (2.033) was more than two times greater than the degrees of freedom, this result was not statistically significant ($p = 0.06$). As $p = 0.06$ is close to the predetermined alpha, the impact of overdispersion has been evaluated. Fewer variables were significant in the Quasi-Poisson model compared to the standard Poisson model, meaning the dispersion parameter does influence the variables' standard errors. In the standard Poisson model, temperature, rainfall, wind speed, traffic and the season variables were significant, whereas in the Quasi-Poisson model only season variables were significant. While the dispersion test suggests that the standard Poisson model should be retained, those variables not significant in the Quasi-Poisson model should be interpreted with caution.

Overall, the diagnostic plots suggested only minor, non-concerning deviance from linearity.

The final log-linked Poisson model included all the original explanatory variables. Average temperature, rainfall, wind speed, traffic and season were all significantly associated with

NZFS incidents. A one degree increase in average temperature was associated with a decrease in the number of NZFS incidents by a factor of 0.79 (21.4%). A one mm increase in average rain was associated with a decrease in the number of NZFS incidents by a factor of 0.82 (18.3%). A one unit increase in average wind speed was associated with an increase in the number of NZFS incidents by a factor of 1.15 (14.7%). A one unit increase in average traffic volume was associated with an increase in the number of NZFS incidents by a factor of 1 (0.04%). When the season is spring, summer or winter compared to autumn, there was an expected decrease in NZFS incidents by 0.02 (98.1%), 0.068 (93.2%) and 0.034 (96.6%).

3.5 Discussion

Our results show that roads can threaten pinnipeds, despite their reduced terrestrial dispersion relative to species typically studied in road ecology.

Between 2012 – 2022, annual average NZFS incidents (59 ±16.9) were almost five times greater than between 1996 – 2005 (12) (Boren et al. 2008). Importantly, 2012 – 2022 incident numbers would likely have been higher still without the effects of the 2016 earthquake and New Zealand's COVID-19 lockdowns. Post-earthquake, it took ca. 17 months for SH1 to reopen in both directions for travel at all hours (Blake et al. 2019), with additional reductions to vehicle flow stemming from traffic slowing measures implemented during the rebuild. Similarly, the longest COVID-19 lockdown affecting the study area saw average daily traffic counts fall by 78.9% between March – April 2020. As tourism recovers post-pandemic, threats to both the NZFS population and motorists will likely increase and need regular assessment. Further, there are economic and societal considerations to this phenomenon, as Kaikōura's economy is heavily reliant on marine mammal tourism (Curtin 2003), creating additional impetus for implementing effective mitigation.

The best explanation for the observed increase in NZFS incidents on SH1 is growth of the local NZFS colonies. Between 2005 – 2015, estimated pup production at Ōhau Point increased from ~600 (Boren et al. 2006b) to 2,471 (Gooday 2016). A 2023 assessment suggests that current pup production is similar to that calculated by Gooday (2016), albeit within a different distribution (Hall et al. 2024 (Chapter 2)). Also, NZFS now breed in additional locations both north and south of the colonies described by Boren et al. (2008) (Figure 3.1) (Hall et al. 2024 (Chapter 2)). Increased average daily traffic volumes, from 2,300 (Boren et al. 2008), to 2,500 (Aghababaei et al. 2020) have likely also contributed, however this increase is small in comparison to the growth in NZFS pup production. Increased road mortality, or, in this case, incidents, due to increased population density has been shown in several species (Rolandsen et al. 2011; Kazemi et al. 2016; Frangini et al. 2022), and the combination of pups' predominance in road incidents, together with the importance of breeding sites in predicting incident locations, supports this explanation. In places where pinniped populations are expanding in proximity to anthropogenic centres and infrastructure, such as South America (Sepúlveda et al. 2020), managers need to be particularly aware of the risks posed by roads, and how these risks can change as the population changes. In New Zealand, the New Zealand sea lion (NZSL; *Phocarctos hookeri*), is an endangered pinniped species also recolonising the country's mainland and may be at particular risk from roads. NZSL can venture up to 1.5 km in land during breeding (Augé et al. 2012), and an adult female and juvenile have already been killed on roads in recent years (Chilvers, pers. comm.). If that species' mainland population continues growing, their interactions with roads should also be closely monitored.

Pups' representation in road mortality increased from the average of 61% recorded between 1996 – 2005 (Boren et al. 2008) to 89.5% for complete years between 2012 – 2022. Notably, Boren et al. (2008) observed that pups' representation in road mortality increased annually through their study, mirroring growth at the Ōhau Point colony, and reaching 83% in 2005. While we did not find a steady increase in pup road mortality, the average is reduced by an anomalously low value in 2015 (58.6%), and, in the final three complete years (2019 – 2021), pups comprised 95.7% of overall road mortality. The age breakdown of animals involved in WVCs is likely species or population specific (Lodé 2000; Borda-de-Água et al. 2014; Garrah et al. 2015), however, over-representation of younger age cohorts has been previously observed in a range of species (Drews 1995; Borda-de-Água et al. 2014; Vyas and Vasava 2019). Potential explanations for this include these individuals' reduced understanding of the dangers of roads (Seiler 2003) and possession of only partially developed sensory and motor functions for judging vehicle speeds and avoiding collisions (Baker et al. 2007). For NZFS, pups largely remain on land between birth and weaning (ca. nine months, Berkson and DeMaster 1985), and are mostly provisioned by their mothers (Harcourt et al. 2002). In road-adjacent colonies, therefore, pups spend more time proximate to roading when compared with weaned individuals, who forage at sea (Costa and Trillmich 1988). This factor, combined with pups' predisposition to exploring novel environments (Gooday 2016), their slower movement (pers. obs.) and the relative novelty of vehicles, may make them more likely to enter a road and less able to avoid vehicles. Some road barriers may also be less effective at excluding pups (see below).

From 2019 (when spatial recording accuracy improved), it was notable that, of the 40 records in which more than one NZFS was recorded in the same location at the same time by NZTA staff, 39 of these records were of pups, all of which had been killed on the road. The unknown lag between death and recording means it is uncertain whether these pups were killed at the same time, however, this could indicate that pups access SH1 in the unstable pods they are known to form when their mothers are at sea (Crawley and Wilson 1976).

The overrepresentation of pups in NZFS incidents demonstrates the importance of going beyond quantifying road incidents, as effective mitigation may depend on understandings of biology and behaviour at different ontogenetic stages (Huijser and Begley 2022). When considering pinnipeds, this result also suggests that proximity to roads may be more problematic for otariids than phocids. Phocids typically have far shorter weaning periods than otariids (Costa and Trillmich 1988) and, as such, even if born near a road, phocid pups would have less opportunity to access it. That said, WVCs involving northern elephant seals (*Mirounga angustirostris*) (Hatfield and Rathbun 1999) demonstrate that road-related risks are still germane to phocids.

Several environmental and temporal variables were associated with changes in the numbers of NZFS incidents. Understanding these drivers can enable temporary mitigation measures to be implemented during periods of heightened risk if permanent solutions are precluded. Increased average wind speeds and traffic volumes were associated with increases in NZFS incidents. The relationship with increasing traffic volumes has been demonstrated in similar studies (Jasińska et al. 2019; Rendall et al. 2021), raising the prospect that, without mitigation, the local problem may become worse as tourism recovers post-pandemic. Further, while this study focused on pinniped colonies near an anthropogenic centre, expanding global road networks, combined with pinniped eco-tourism (Granquist and Sigurjonsdottir 2014), could mean geographically isolated colonies come under greater threat.

Relationships between wind speed and WVCs are less well-established, particularly with regards to mammals. Perhaps unsurprisingly wind speed has been shown to influence WVC mortality rates in birds, driving both increases (Erritzoe et al. 2003) and decreases (Soares and Dias 2020) in mortality rates among different species. Increased wind speeds have been positively associated with the numbers of Australian fur seals (*Arctocephalus pusillus doriferus*) ashore (Garlepp et al. 2014), and NZFS move inland during storms (McNab and

Crawley 1975). As such, stronger winds could cause NZFS near SH1 to access the road to escape deteriorating ocean conditions. Conversely, increased average temperature and rainfall were associated with declining NZFS incidents numbers. Road-related responses to temperature and rainfall are likely species specific (D'Amico et al. 2015; Nelli et al. 2018; Raymond et al. 2021), and in NZFS, these relationships are likely based on finding shelter. Young pups are particularly vulnerable to extreme heat (De Villiers and Roux 1992), and lack water repellent fur (Erdsack et al. 2013), and so would likely avoid exposed road surfaces if confronted with high temperatures or heavy/prolonged rain.

Of the environmental and temporal variables, season had the greatest effect on incidents, and was the sole significant variable returned by the Quasi-Poisson model. Autumn (March – May) had significantly more NZFS incidents than any other season, and, given pups' predominance in NZFS incidents, the most likely explanation for this is pup ontogeny. During summer, once born, pups are initially relatively immobile (Crawley and Wilson 1976), however, by the start of autumn, they are three to four months old and more mobile and inquisitive (Gooday 2016) They are also left onshore by their mothers for more extended periods, therefore suckling less frequently (5 to 7 days (Crawley and Wilson 1976)). Combined, these factors are the likely reasons why autumn is particularly dangerous for pups born in road-adjacent colonies. By winter, pups are preparing for weaning (Baylis et al. 2005), and thus spend more time learning to hunt at sea for themselves, and by spring they are weaned and are thus less consistently at the colony (Stirling 1971). While our strongest mitigation recommendations involve constructing permanent barriers on SH1, these results indicate that temporary barriers at hotspots in autumn, or during high winds or times of increased traffic volumes, could represent effective alternatives. Given that many coastal and some marine species (such as sea turtles) exhibit high seasonal variation in when they are ashore (Higham 1998; Schunck et al. 2023; Sella et al. 2023), seasonal analyses should be prioritised for understanding the risks posed by roads to these species.

The contemporary hotspot analysis identified significant clusters both north and south of Kaikōura. Notably, there were no significant clusters within the boundaries of the pre-earthquake Ōhau Point colony, which previously contained the region's densest aggregation of NZFS (Boren et al. 2006b). From our results, effectively mitigating just 2.75 km of the 90km of SH1 in this region could prevent almost 90% of NZFS incidents.

The KDE+ method, like other kernel density estimation-based methods (Xie and Yan 2008; Anderson 2009) permits more precise location of clusters than traditional methods involving Poisson or negative binomial approaches (e.g. Lee et al. 2006; Malo et al. 2004) (Bíl et al. 2019). The latter involve aggregating incidents into arbitrarily defined road segments, and thus can only define those segments as significant or not. Additionally, methods involving arbitrary segmentation could inadvertently separate incidents within a cluster. By contrast, the KDE+ method uses the locations of incidents to define the starts and ends of significant clusters. Further, the KDE+ method's capacity to rank significant clusters by strength enables the prioritisation of the most important sites for mitigation. This is particularly beneficial when assessing large mammal WVCs, as effective mitigation is often costly (Rytwinski et al. 2016). The method's targeting capabilities are taken a step further with the more stringent 'segment test', which is applied to each road segment in its entirety, rather than to every location on the road. This can reduce false alarm rates when identifying the most concerning locations. The four segments identified by the segment test here accounted for only 1.95 km of SH1 and contained 60.4% of the NZFS incidents. While we strongly recommend permanent mitigation measures be implemented at all the significant sites highlighted in Figure 3.2, if resources were limited, the sections that met the threshold of the segment test should be prioritised.

The presence of a road abutting breeding area was by far the most important driver of the spatial variation in NZFS incidents, correctly classifying 74 out of 78 observations. Breeding sites contain the greatest year-round NZFS density, and increased population density has been associated with increased WVC rates in several species (Rolandsen et al. 2011; Kazemi et al. 2016). Additionally, NZFS pups (the predominant age group in road incidents) typically remain in the vicinity of their birth site until they wean (Crawley and Wilson 1976). The importance of breeding sites in predicting incident locations has wider implications when targeting road incident mitigation for fur seals. While substantial numbers of adult and sub-adult males haul out away from rookeries outside the breeding season (Crawley and Wilson 1976), including at road-adjacent sites along the Kaikōura coast (pers obs.), such sites appear less problematic than rookeries from the perspective of WVCs. However, as adult haul-out sites can develop into breeding areas as part of the fur seal recolonization process (Dix 1993; Ryan et al. 1997), monitoring of road incidents near haul out areas remains important.

The hotspot persistence analyses demonstrated clear shifts in the epicentre of significant kilometre sections over time, from the north to the south of the study area. The increasing prevalence of incidents south of Kaikōura correlates with population growth in this area. Pre-earthquake, there were no published records of mainland NZFS breeding south of Kaikōura within the study area (Boren et al. 2006b), whereas a 2023 assessment identified multiple breeding sites along this coastline (Hall et al. 2024 (Chapter 2)). However, pup production remains higher in the north coast colonies (Hall et al. 2024 (Chapter 2)), and, given the links between population density and WVCs (Rolandsen et al. 2011; Kazemi et al. 2016; Frangini et al. 2022), it would be expected that this area would remain the epicentre of local NZFS incidents.

The most significant changes to SH1's infrastructure since the earthquake took place north of Kaikōura. This included adding 2.5 km of non-contiguous sea wall, most of which is around

Ōhau Point, the largest NZFS colony, and stands between 8-10 metres above sea level. Segments now containing sea wall, such as 118.12 and 118.13, have shown decreases in overall NZFS incident numbers since this barrier's construction in 2018 (Table 3.5). By using historic RoadRunner videos, it is also possible to approximate the timing of modifications to other road barriers and compare this to trends in NZFS incidents. For example, in segments such as 118.11 (Figure 3.3a), motorcycle protection rail (MPR – double guard rail) implementation coincided with substantial decreases to NZFS incidents. By contrast, in south coast segments, such as 163.06 (Figure 3.3b), large increases in NZFS incidents have occurred concomitantly with the erection of single guard rail and plastic mesh safety fence (PMSF), suggesting that these barriers do not effectively exclude NZFS.

Our results, therefore, support evidence that changes to roading infrastructure can modify impacts on proximate wildlife (Ciocheti et al. 2017; Secco et al. 2022). Such impacts appear to be variable, largely due to differences in species behaviour and habitat preferences (Ciocheti et al. 2017). Generally, post-earthquake modifications to SH1 appear to have reduced NZFS incidents in previous hotspots north of Kaikōura. However, the road is now closer to the coast, which has reduced the depth of some NZFS colonies (pers. obs.), underlining the complex range of effects that must be considered when planning and implementing a wildlife-adjacent infrastructure project.

These results also demonstrate why species behaviour and biology must be considered when selecting exclusion barriers (Woltz et al. 2008). Given the small size of NZFS pups, which are over-represented in NZFS incidents on SH1, it is unlikely that a barrier like single guard rail, where the lowest edge of the railing stands more than 40cm above the road surface, would effectively exclude them. The most effective barrier for excluding NZFS pups would, therefore, likely be one that reaches the road surface. Such measures have been effective in preventing road access in other species with short limbs and a limited agility, such as turtles

(Woltz et al. 2008). Barriers fitting this description currently exist on the Kaikōura coast and experience varying success. For example, MPR appears to have reduced the number of incidents in parts of 118.11 compared with the previous barrier in place in this section – metal wire fencing that did not reach the road surface. Contrastingly, PMSF in 163.06 and 163.07 appears ineffective, as incident numbers there have increased dramatically despite this barrier being extended. This is likely due to a combination of poor maintenance (pers. obs.), and fence-end effects (Huijser et al. 2016; Kim et al. 2021). For example, in 163.06, the two PMSF sections do not stretch the entire length of their nearest breeding areas, so it is not surprising that this section experiences high NZFS incident numbers. We recommend that, at a minimum, barriers should span the entire length of the nearest breeding sites. As this study has demonstrated that hotspot locations may shift over time, it will also be necessary to continue the current monitoring approaches, and update mitigation responses as necessary. Other actions that could reduce the occurrence of NZFS incidents include reducing maximum vehicle speeds in high-risk areas and increasing the amount of warning signs regarding the possibility of NZFS on the roads to increase driver awareness.

The issues highlighted here are likely to be replicated elsewhere in New Zealand as NZFS continue to recolonise areas within their pre-exploitation range (Dix 1993; Bouma et al. 2008; Cowling et al. 2014), and in other countries where pinnipeds are recovering from exploitation (Kirkwood et al. 2010; Milano et al. 2020b). The problem is also likely to be exacerbated by the synergistic climate-change induced impacts of sea level rise and increasingly frequent and severe storms, which may force coastal species such as fur seals to retreat inland (McLean et al. 2018).

3.6 Conclusions

This research demonstrates how roads can impact coastal species, which are typically underrepresented in road ecology studies, and highlights the benefits of identifying WVC hotspots to permit the most efficient allocation of mitigation resources. As such, hotspot analyses could be conducted for any wildlife populations impacted by road mortality, or the threat thereof.

Where wildlife populations are threatened by road mortality, species-specific barriers should be selected based on what will be most effective for reducing WVCs. At locations in which multiple species are threatened, barrier design should account for differences in their behaviour and ontogeny. Additionally, predictive habitat studies are required for wildlife populations that are expanding near roads to allow assessments of probable areas of future habitation, and thus enable proactive decision making regarding future exclusion measures (MacMillan et al. 2016).

Chapter 4 Planning for a pinniped response during a marine oil spill

Bull New Zealand fur seal in Sleepy Bay, Banks Peninsula

This chapter is a reformatted version of the following manuscript, currently in press with Environmental Science and Pollution Research.

Hall, A. A., Chilvers, B. L. & Weir, J. S. (*in press*). Planning for a pinniped response during a marine oil spill.

MASSEY UNIVERSITY
TE KUNENGA KI PŪREHUROA
UNIVERSITY OF NEW ZEALAND

GRADUATE RESEARCH SCHOOL

STATEMENT OF CONTRIBUTION
DOCTORATE WITH PUBLICATIONS/MANUSCRIPTS

We, the student and the student's main supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the student's contribution as indicated below in the Statement of Originality.

Student name:	Alasdair Hall
Name and title of main supervisor:	Prof. Louise Chilvers
In which chapter is the manuscript/published work?	4

Describe the contribution that the student and members of the supervisory team have made to the manuscript/published work:[1]

Alasdair Hall, Louise Chilvers and Jody Weir contributed to the study conception and methodology design. Alasdair Hall and Louise Chilvers contributed to funding acquisition, data collection, data analysis and visualisation. The first draft of the manuscript was written by Alasdair Hall and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Please select one of the following three options:

- [x] **The manuscript/published work is published or in press**

  Please provide the full reference of the research output:

  Hall, A. A., Chilvers, B. L. & Weir, J. S. (in press). Planning for a pinniped response during a marine oil spill. Environmental Science and Pollution Research.

- [ ] **The manuscript is currently under review for publication**

  Please provide the name of the journal:

- [ ] **It is intended that the manuscript will be published, but it has not yet been submitted to a journal**

Student's signature:	Alasdair Hall Digitally signed by Alasdair Hall Date: 2025.03.16 14:33:11 +13'00'	Main supervisor's signature:	Barbara Louise Chilvers Digitally signed by Barbara Louise Chilvers DN: cn=Barbara Louise Chilvers, c=NZ, o=Massey University, ou=Wildbase, School of Veterinary Science, email=b.l.chilvers@massey.ac.nz Date: 2025.03.17 09:56:19 +13'00'

This form should be placed at the beginning of each relevant thesis chapter.

[1] Refer to the Massey University Publishing and Authorship guidelines (OneMassey for staff, Stream for students) and/ or Contributor Roles Taxonomy (CRediT) guidelines for guidance.

4.1 Abstract

Understanding the distribution and abundance of wildlife populations is key to successful oil spill response planning. Fur seals are difficult to rehabilitate if oiled, and many common spill response techniques may be limited in the high-energy rocky shore habitats they prefer. Preventing oil from reaching colonies and hazing or deterring animals away from oil are high priority response options for pinnipeds during spills. To do this, local knowledge of pinniped distribution and abundance is required, as well as knowledge of effective and safe hazing and deterrence mechanisms. From pup production assessments, we estimated that a population of 13,147 – 17,675 New Zealand fur seal ('NZFS': *Arctocephalus forsteri*) currently inhabits Banks Peninsula. This area contains the largest port on New Zealand's South Island and a secondary port that is popular with cruise ships, elevating its oil spill risk profile. From the knowledge gained regarding NZFS distribution and abundance, we evaluated mitigation methods which could protect fur seals during oil spills, wherever these species occur, and make suggestions to managers on how to mount an effective pinniped response.

4.2 Introduction

Acute oil spills are among the most highly publicised threats to marine environments and the animals that exist within them (Fan et al. 2015). Given the many deleterious impacts of marine oil spills, and public expectations of a response to oiled wildlife (Safford et al. 2012; Henkel and Ziccardi 2018), many countries now require companies involved in oil acquisition and transportation to produce response plans that include consideration of oiled wildlife (Chilvers et al. 2021). In such plans, the optimum wildlife response is preventing the oil from reaching animals (Wolfaardt et al. 2008; Nijkamp et al. 2014; Ziccardi et al. 2015; Hong et al. 2020). This necessitates rapid mobilisation of personnel and resources and requires that baseline ecological data are in place (Battershill et al. 2016) regarding what species are likely to be present, where they are likely to be, and their local abundance. Even if the prevention objective cannot be met, knowledge of species' pre-spill distribution and abundance is key to determining the incident's ecological impact and informing subsequent management (Lewis et al. 2020; Fraser et al. 2022).

New Zealand's most significant oil spill was the grounding of the *MV Rena* off Tauranga in 2011 (Schiel et al. 2016), with an estimated minimum of ca. 2,500 seabirds, 24 terrestrial birds, 17 kekeno/New Zealand fur seal (NZFS; *Arctocephalus forsteri*) and three whales oiled as a result (Sievwright 2014). One of the recommendations derived from studies of this incident was that ecological data should be collected for areas where spills are most likely to occur in New Zealand, such as near ports and drilling platforms (Battershill et al. 2016).

Maritime NZ, the government authority responsible for responding to significant oil spills in New Zealand, identified Lyttelton Harbour on Banks Peninsula (Figure 4.1), as falling into a high-risk classification category for oil spills (Navigatus Consulting 2015). Lyttelton is the South Island's major port, and one of the busiest in the country (Inglis et al. 2008). Further, the recent closure of New Zealand's only oil refinery means the country is now totally reliant on oil imports from overseas (Wilson et al. 2023), and important ports such as Lyttelton will experience greater numbers of oil transportation vessels, heightening local spill risk (Chilvers et al. 2021). Banks Peninsula is an ecologically important area, home to threatened, endangered and rare species including Hector's dolphins (*Cephalorhynchus hectori*) (Carome et al. 2022) spotted shags (*Stictocarbo punctatus*) (Brough et al. 2019) and white-flippered penguins (*Eudyptula minor albosignata*) (Allen et al. 2011), as well as NZFS (Ryan et al. 1997; Bradshaw et al. 1999; Boren et al. 2006b). The risk of oil spills around Banks Peninsula is heightened by the proximity of a second port at Akaroa (Figure 4.1). Akaroa is not a commercial port but is popular with large cruise boats (Carome et al. 2022), and Banks Peninsula is also important for commercial fishing. Shipping is the most common source for acute marine oil spills (Chilvers et al. 2021), meaning increased vessel traffic raises the overall risk profile for Banks Peninsula.

Fur seals (the nine *Arctocephalus* spp. and the northern fur seal *(Callorhinus ursinus*)), are at particular risk from oil spills due to their biology and life history (Mearns et al. 1999; Helm et al. 2015). Fur seals rely on their fur for insulation, and oil fouling significantly hinders their thermoregulatory capabilities (Mearns et al. 1999; Helm et al. 2015), reducing their ability to swim and forage (Ziccardi et al. 2015). This contrasts with all other pinnipeds, including phocids (true seals), odobenids (walrus) and sea lions which rely on blubber for insulation and so do not have the same dense, hard-to-clean fur (Ziccardi et al. 2015). The typical high density of fur seal colonies also means that proximal oil spills have the capacity to impact large numbers of animals at once, demonstrated by the deaths of at least 4,738 South American fur seal (*Arctocephalus australis*) pups following the *San Jorge* spill off Uruguay (Mearns et al. 1999). Large mammals, like fur seals, are also highly challenging to rescue and rehabilitate (Northwest Area Committee 2019). Often, only certain age classes can be considered for treatment if oiled, as the size and aggressiveness of sub-adults and adults (particularly males) means that response options beyond monitoring are likely to be limited (Chilvers 2017). The *San Jorge* spill demonstrated the difficulty of treating even the smallest fur seals age cohorts, as all 41 pups taken for cleaning ultimately died (Mearns et al. 1999). Except for New Zealand sea lion (*Phocarctos hookeri*) pups, and potentially some juveniles or small females of this species, there would be limited ability to treat other, transient pinnipeds that occur in New Zealand – southern elephant seals (*Mirounga leonina*) due to their size, and leopard seals (*Hydrurga leptonyx*) due to their aggressiveness. New Zealand also lacks permanent rehabilitation facilities capable of treating large numbers of oiled seals. The only facility in New Zealand that currently holds pinnipeds in captivity is Auckland Zoo, and transporting NZFS large distances to this centre, near the top of the North Island, would be impractical. As such, in New Zealand, like many countries, other response approaches, primarily aimed at preventing pinnipeds from becoming oiled, need to be explored and prioritised.

Despite the need for baseline ecological data in high spill risk areas (Battershill et al. 2016), NZFS' high oiling likelihood (Chilvers 2017) and the presence of two ports on Banks Peninsula, there are no reliable contemporary distribution and abundance data for NZFS in this region. This is representative of much of New Zealand, where NZFS population monitoring has been *ad hoc* with regards to regularity and methodological approach (Dix 1993; Boren et al. 2006b; Bouma et al. 2008; Gooday and Goldstien 2018; Chilvers 2021), as the species continues to recolonise areas within its former range post historic sealing (Dussex et al. 2016). Currently, no reliable nationwide abundance estimate for NZFS exists, with a figure of 200,000 being used for the combined population of NZFS in New Zealand and Australia for over 20 years (Harcourt 2001; Goldsworthy and Gales 2008). The recovery of fur seal species from historic exploitation has been observed in several countries (Kirkwood et al. 2010; Milano et al. 2020b) and has often followed a documented pattern (Roux 1987), knowledge of which can inform oil spill response planning (see Discussion). On Banks Peninsula, the first post-sealing record of NZFS breeding was from Horseshoe Bay in 1973 (Wilson 1981), and the most recent thorough assessment of individual colonies was ca. 20 years ago, when pup production estimates of ca. 300 were calculated for both Horseshoe Bay and Te Oka Bay (Boren et al. 2006b).

This study aims to provide current understandings of NZFS abundance and breeding distribution on Banks Peninsula to facilitate regional oil spill response planning and species management. Based on the findings, we also discuss and evaluate possible mitigation and remediation options, to improve future pinniped responses during marine oil spills. These recommendations can be adapted for any country where fur seals are present.

4.3 Materials and methods

Banks Peninsula is a peninsula of ca. 1,150 square kilometres, located centrally on the east coast of New Zealand's South Island (Figure 4.1).

In this study, locations of large aggregations of NZFS around Banks Peninsula were initially noted during land and boat-based surveys in October 2023. NZFS colony sizes on Banks Peninsula were determined through pup direct count and mark-recapture surveys between January 30th and February 29th, 2024. Pup production estimates are the most reliable indicators for total pinniped population sizes (Berkson and DeMaster 1985), and late January – March represents the period when the entire year's cohort of NZFS pups will have been born, and thus are available to count, and still retain their instantly recognisable black natal pelage (Berkson and DeMaster 1985). Mark-recapture and direct counts of pups have been used to provide NZFS abundance estimates for several decades (Taylor et al. 1995; Bradshaw et al. 2003; Boren et al. 2006b; Roberts and Neale 2016), and have recently been used to provide updates to regional populations that had gone unstudied for some time (Chilvers 2021; Hall et al. 2024 (Chapter 2)). Here, a 'colony' refers to a breeding aggregation of NZFS, after Shaughnessy et al. (2015).

Most surveys were conducted from the land, but five sites (Figure 4.1: Nos. 11 (west arm), 17, 18, 21, 32) were assessed from vessels due to a lack of terrestrial access. Survey effort distribution is shown in Figure 4.1.

Terrestrial direct counts involved observers moving systematically through colonies and counting every NZFS pup they could find. In some locations, where access to breeding was unfeasible or unsafe, counts were conducted from a vantage point such as a clifftop.

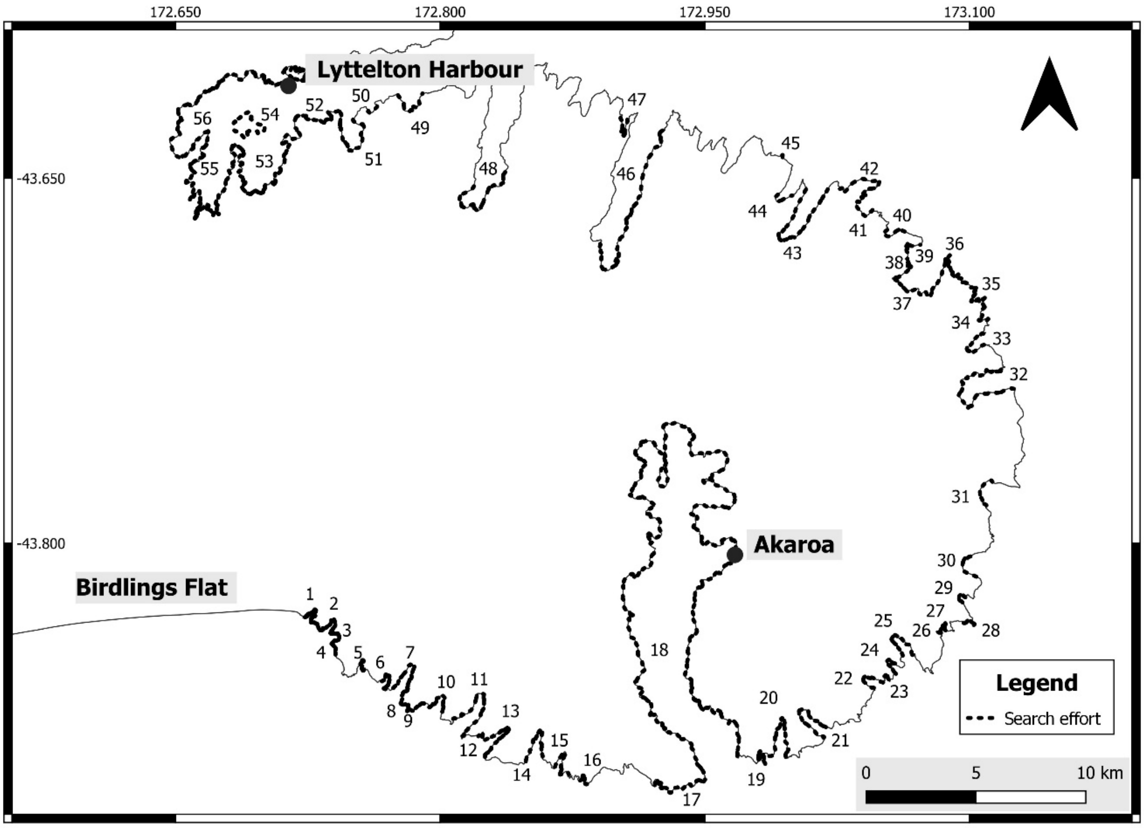


Figure 4.1 Key locations within the study area and distribution of search effort. The names of the bays are shown as numbers and correspond as follows: 1. Oashore Bay, 2. Tokoroa Bay, 3. Hikuraki Bay, 4. Magnet Bay beach, 5. Murrays Mistake, 6. Tumbledown Bay, 7. Te Oka Bay, 8. Te Kaio Bay, 9. Hells Gate, 10. Robin Hood Bay, 11. Peraki Bay, 12. Unnamed Bay, 13. Horseshoe Bay, 14. Long Bay, 15. Island Bay, 16. Whakamoa Bay, 17. Scenery Nook, 18. Akaroa Harbour, 19. Haylocks Bay, 20. Damons Bay, 21. Pōhatu Bay, 22. Stony Bay, 23. Reef Nook, 24. Sleepy Bay, 25. Otanerito Bay, 26. Red Bay, 27. Shell Bay, 28.Goat Point, 29. Paua Bay, 30. Goughs Bay, 31. Hickory Bay, 32. Le Bons Bay, 33. Lavericks Bay, 34. Ducksfoot Bay, 35. Pā Bay, 36. East Head, 37. Okains Bay, 38. Spyglass Point, 39. North West Bay, 40. Stony Bay, 41. Raupo Bay, 42. Long Lookout Point, 43. Little Akaloa Bay, 44. Decanter Bay, 45.South of Squally Bay, 46. Pigeon Bay, 47. Little Pigeon Bay, 48. Port Levy/Koukourarata, 49. Camp Bay, 50. Pile Bay, 51. Purau Bay, 52. Diamond Harbour, 53. Charteris Bay, 54. Otamahua/Quail Island, 55. Head of the Bay, 56. Governors Bay

Mark-recapture was performed at colonies where pup numbers were thought to be over 100 from preliminary assessments (Boren et al. 2006b; Roberts and Neale 2016; Chilvers 2021; Hall et al. 2024 (Chapter 2)). In three instances (Pā Bay, Peraki Bay and Island Bay; Table 4.3) this was not possible due to a lack of access to pups, meaning direct counts were performed instead. Mark-recapture was favoured at sites with more pups as this technique produces higher estimates than direct counts, as, in the latter, some pups may be concealed from the counters' view (Watson et al. 2009). In both methodologies, dead pups were tallied and added to the live counts.

During mark-recapture, pups were marked non-permanently by trimming a patch of natal fur on their heads (Boren et al. 2006b; Chilvers 2021; Hall et al. 2024 (Chapter 2)). Recaptures took place the day after marking and involved observers systematically traversing the colonies, recording the numbers of marked and unmarked pups (Boren et al. 2006b; Chilvers 2021).

The ratio of marked to unmarked pups in each recapture sample were input into a modified Petersen pup production estimate (Chapman 1952) as follows:

$$Pi = \left[\frac{(M+1)(Ci+1)}{Ri+1}\right] - 1$$

Here, for each replicate i, M is the number of marked NZFS pups, C_i represents the total number of pups in the recapture sample and R_i is the number of marked pups in the recapture sample. Subsequently, the grand total estimate of pup production, P, is calculated from the mean of the Q Petersen estimators as:

$$\frac{\sum_{i=1}^{Q} Pi}{Q}$$

As direct counts are known to produce underestimates (Watson et al. 2009), wherever mark-recapture was undertaken, a direct count was also conducted so that calibration indices could be calculated. These were used to convert the results from direct count only sites into results comparable with the mark-recapture results (Watson et al. 2009; Chilvers 2021; Hall et al. 2024 (Chapter 2)).

Two estimates of pup production were derived from these analyses. A minimum pup production estimate was calculated by summing the mark-recapture results and the direct count only sites. Second, a converted pup estimate was calculated by multiplying the results of direct count only sites by a calibration index and adding these results to the sum of the mark-recapture estimates.

Standard errors for mark-recaptures and direct counts were calculated after Gales and Fletcher (1999).

Multipliers have previously been used to convert pup production estimates into population estimates (Shaughnessy et al. 1994, 1995; Campbell et al. 2014; Chilvers 2021; Hall et al. 2024 (Chapter 2)). Here, Goldsworthy and Page's (2007) multiplication factor of 4.76 was used as this is the most recently calculated index and uses NZFS life history data, as opposed to other that from species.

In New Zealand, oil spill responses are coordinated by either the regional council in the impacted area, or at a national level by Maritime New Zealand (Maritime NZ), depending on the spill size and complexity. There is also a National Oiled Wildlife Response Team run out of Massey University, with trained responders located throughout the country. Maritime NZ publishes a periodically reviewed Oil Spill Readiness and Response Strategy (Maritime NZ 2021), which details the systems and procedures in place for responding to major oil spills that go beyond the capacity of regional councils. Regional councils create similar documents for coordinating responses in their regions. Environment Canterbury (ECAN) is the regional council responsible for responding to oil spills impacting Banks Peninsula. If a spill were to occur around Banks Peninsula, depending on its exact location, the closest available spill kits are in Akaroa or Lyttelton, and there are approximately 30 people trained to respond to a spill in the region (Environment Canterbury 2024).

4.4 Results

The distribution of NZFS breeding around Banks Peninsula is shown in Figures 4.2 – 4.4. A total of 41 colonies were identified, 25 of which had not been surveyed before. South of Okains Bay (Figures 4.1 & 4.4), pupping occurred at all searched sites other than Akaroa Harbour, and the headland directly south-west of the Akaroa Harbour mouth. NZFS pupping north of Okains Bay was less consistent relative to search effort. No NZFS pups were recorded west of Squally Bay on the north coast of Banks Peninsula, or within Akaroa Harbour (Figures 4.1 & 4.4).

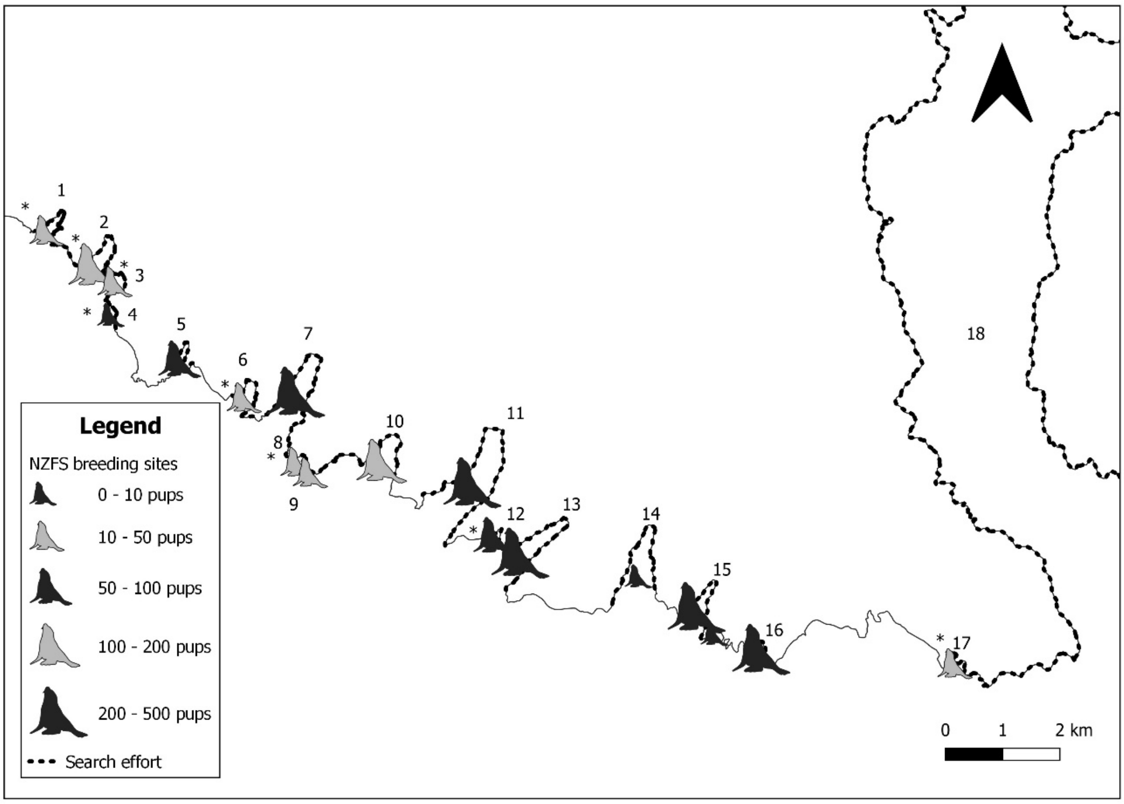


Figure 4.2 Locations of NZFS colonies on the south-west coast of Banks Peninsula. Colonies marked with * are sites where there is no previous record of NZFS breeding (Baird 2011; Emami-Khoyi et al. 2016a). The names of the bays are shown as numbers and correspond as follows: 1. Oashore Bay, 2. Tokoroa Bay, 3. Hikuraki Bay, 4. Magnet Bay beach, 5. Murrays Mistake, 6. Tumbledown Bay, 7. Te Oka Bay, 8. Te Kaio Bay, 9. Hells Gate, 10. Robin Hood Bay, 11. Peraki Bay, 12. Unnamed Bay, 13. Horseshoe Bay, 14. Long Bay, 15. Island Bay, 16. Whakamoa Bay, 17. Scenery Nook, 18. Akaroa Harbour

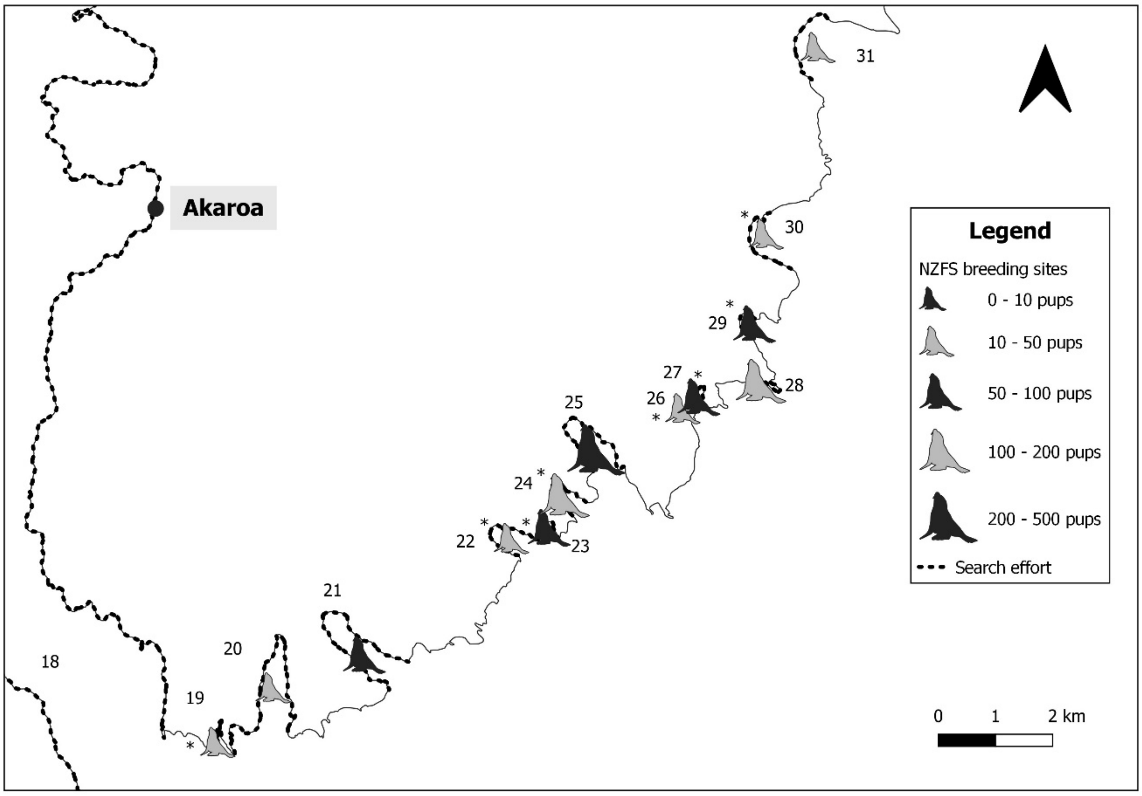


Figure 4.3 Locations of NZFS colonies on the south-eastern coast of Banks Peninsula. Colonies marked with * are sites where there is no previous record of NZFS breeding (Baird 2011; Emami-Khoyi et al. 2016a). The names of the bays are shown as numbers and correspond as follows: 18. Akaroa Harbour, 19. Haylocks Bay, 20. Damons Bay, 21. Pōhatu Bay, 22. Stony Bay, 23. Reef Nook, 24. Sleepy Bay, 25. Otanerito Bay, 26. Red Bay, 27. Shell Bay, 28. Goat Point, 29. Paua Bay, 30. Goughs Bay, 31. Hickory Bay

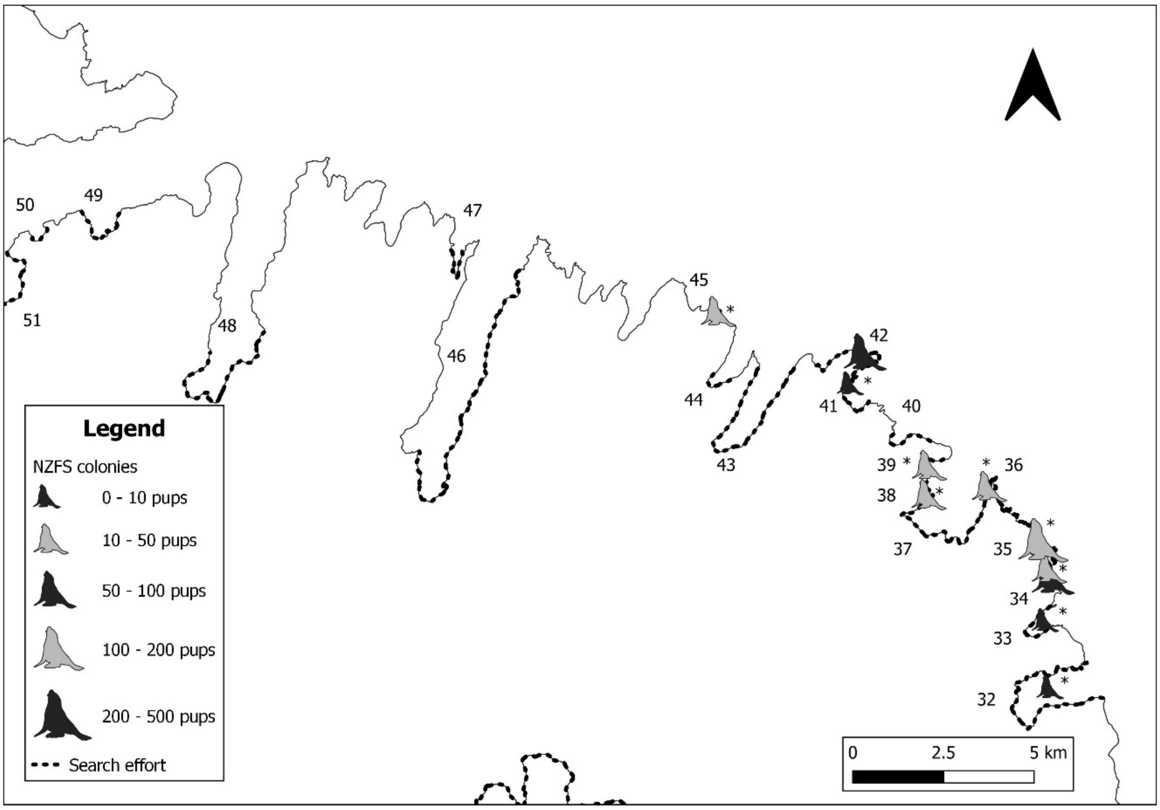


Figure 4.4 Locations of NZFS colonies on the north-eastern coast of Banks Peninsula. Colonies marked with * are sites where there is no previous record of NZFS breeding (Baird 2011; Emami-Khoyi et al. 2016a). The names of the bays are shown as numbers and correspond as follows: 32. Le Bons Bay, 33. Lavericks Bay, 34. Ducksfoot Bay, 35. Pā Bay, 36. East Head, 37. Okains Bay, 38. Spyglass Point, 39. North West Bay, 40. Stony Bay, 41. Raupo Bay, 42. Long Lookout Point, 43. Little Akaloa Bay, 44. Decanter Bay, 45.Squally Bay, 46. Pigeon Bay, 47. Little Pigeon Bay, 48. Port Levy/Koukourarata, 49. Camp Bay, 50. Pile Bay, 51. Purau Bay

A total of 267 pups were marked during mark-recapture across five sites (Table 4.1), producing an estimated pup production of 1,321 (± 15 SE). Direct count only sites (Table 4.3) produced a total estimated pup production of 1,451 (± 2 SE). The results from the mark-recapture sites and direct count only sites were summed to produce a minimum NZFS pup production estimate for Banks Peninsula of 2,762 – 2,774 pups in the 2023/24 breeding season.

Table 4.1 Numbers of New Zealand fur seal pups marked at sites on Banks Peninsula

	Pups marked	
Site	**Number**	**Percentage of site total (%)**
Te Oka Bay – east	70	20.47
Horseshoe Bay – east	60	15.74
Whakamoa Bay	65	25.49
Otanerito Bay – north	37	16.9
Goat Point	35	28.46
Total	267	

Five calibration indices were calculated from the sites where both mark-recapture and direct counts were conducted (Table 4.2).

Table 4.2 Comparison of direct count and mark-recapture results at sites where both were conducted to show derivation of calibration indices

Site	Direct count result	Mark-recapture result	Calibration index
Te Oka Bay – east	186	333	1.80
Horseshoe Bay – east	185	377	2.04
Whakamoa Bay	194	255	1.31
Otanerito – north	115	215	1.88
Goat Point	107	124	1.16

Five indices were calculated because Banks Peninsula demonstrates substantial variation in its topography, making a single calibration index unlikely to be appropriate for use at all sites (Watson et al. 2009). Calibration index selection was based on matching most closely the physical characteristics of the bay in question (such as slope steepness and degree of cliff overhang) that could impede the counter's view of the pups, with one for which a calibration index existed (Table 4.3). When direct count only sites were calibrated and added to the mark-recapture estimates, the converted pup production estimate for Banks Peninsula was 3,702 – 3,713.

By subjecting the lower bound of the minimum pup production estimate (2,762) and the upper bound of the converted pup production estimate (3,713) to Goldsworthy and Page's (2007) 4.76 multiplier (Chilvers 2021; Hall et al. 2024 (Chapter 2)), a total NZFS population estimate for Banks Peninsula in the 2023/24 breeding season was calculated as 13,147 – 17,675.

Table 4.3 New Zealand fur seal pup production estimates for Banks Peninsula in the 2023/24 breeding season

Site	Colony reference no. (Figs. 4.1 – 4.4)	Pup direct count (mean ± SE)	Pup mark-recapture estimate (mean ± SE)	Converted bay total (mean ± SE)
Oashore*	1	27		48
Tokoroa*	2	84 ± 3		151 ± 3
Hikuraki*	3	10 ± 1		18 ± 1
Magnet Bay Beach*	4	1		2
Murray's Mistake*	5	46 ± 3		83 ± 3
Tumbledown*	6	17		31
Te Oka – west* Te Oka – east	7	42 ± 1 195 ± 8	342	418 ± 1
Te Kaio*	8	10		18
Hells Gate*	9	14		26
Robin Hood Bay**	10	60 ± 1		123 ± 1
Peraki Bay**	11	153 ± 5		312 ± 5
Unnamed bay between Peraki and Horseshoe**	12	47 ± 2		96 ± 2

Horseshoe Bay – west**	13	51± 1		485 ± 20
Horseshoe Bay – east		189 ± 14	381 ± 18	
Long Bay**	14	1		2
Island Bay**	15	101 ± 5		206 ± 5
Headland between Island Bay and Whakamoa Bay***	Between 15 and 16	6 ± 2		7 ± 2
Whakamoa Bay – west***	16	99 ± 14	119 ± 5	255 ± 12
Whakamoa Bay – east***		95 ± 9	135 ± 11	
Scenery Nook***	17	30		39
Haylocks Bay***	19	12		16
Damons Bay****	20	14		26
Pōhatu Bay****	21	45		85
Stony Bay****	22	15 ± 1		28 ± 1
Reef Nook****	23	28 ± 1		53 ± 1
Sleepy Bay****	24	70 ± 1		132 ± 1
Otanerito Bay – south ****	25	12 ± 1		241 ± 16
Otanerito Bay – north		119 ± 3	219 ± 9	
Red Bay****	26	15 ± 1		28 ± 1
Shell Bay****	27	37 ± 2		69 ± 2
Goat Point*****	28	107 ± 7	124 ± 4	124 ± 4
Paua Bay*****	29	68 ± 6		79 ± 6
Goughs Bay*****	30	12 ± 1		14 ± 1
Hickory Bay*****	31	28 ± 2		32 ± 2
Le Bons Bay*****	32	9		10
Lavericks Bay*****	33	6		7
Ducksfoot Bay*****	34	84 ± 1		97 ± 1
Headland between	Between 34 and 35	18		21

Ducksfoot Bay and Pā Bay*****				
Pā Bay*****	35	130 ± 4		150 ± 4
Okains Bay – East Head*****	36	25 ± 2		29 ± 2
Okains Bay – Spyglass Point*****	38	32 ± 1		37 ± 1
North West Bay*****	39	14		16
Raupo Bay*****	41	7		8
Long Lookout Point*****	42	65 ± 1		75 ± 1
East of Squally Bay*****	Between 44 and 45	10		12
Total		**1,451 ± 2[a]**	**1,321 ±15**	
Minimum pup estimate [b]		**2,762 – 2,774**		
Converted pup estimate [c]			**3,702 – 3,713**	
Coarse estimated total population range (multiplier 4.76) [d]		**13,147 – 17,675**		

Note: Empty cells indicate no count of that type was conducted in that location.

[a] This total is the sum of the direct counts for the sites where mark–recapture was not also undertaken.

[b] The minimum pup estimate is the sum of the direct counts for the sites where mark–recapture was not undertaken, and the mark-recapture results from the remaining sites.

[c] The converted pup estimate is the sum of the mark-recapture estimates combined with the sum of the products of each direct count multiplied by the given calibration index.

[d] This population estimate equals the range between the lower bound of the minimum pup estimate multiplied by 4.76 and the upper bound of the converted pup estimate multiplied by 4.76 (Goldsworthy and Page 2007).

* Te Oka Bay multiplier used to convert direct counts.

** Horseshoe Bay multiplier used to convert direct counts.

*** Whakamoa Bay multiplier used to convert direct counts.

**** Otanerito Bay multiplier used to convert direct counts.

***** Goat Point multiplier used to convert direct counts.

4.5 Discussion

Knowledge of species distribution and abundance is important not only for oil spill response planning (Battershill et al. 2016), but also for determining the short and long-term ecological impacts of oil spills (Lewis et al. 2020; Fraser et al. 2022), or other unexpected events such as natural disasters (Hall et al. 2024 (Chapter 2)). This study sought to provide important population parameters for NZFS on Banks Peninsula, in the first comprehensive survey of the region to aid with oil spill response planning. In the 2023/24 breeding season, NZFS breeding was identified at 41 bays or headlands around Banks Peninsula, 25 of which had not previously been recorded (Figures 4.2 – 4.4; Table 4.3). Using a combination of mark-recapture and direct counts, a minimum pup production estimate of 2,762 – 2,774 was produced. As direct counts typically produce underestimates (Watson et al. 2009), calibration indices were used to produce a more reliable pup production estimate of 3,702 – 3,713. The total NZFS population on Banks Peninsula in 2023/24 was estimated at 13,147 – 17,675. This figure is likely still an underestimate, as five colonies where guesstimates of between 20 – 50 pups had been recorded in 2007 (Baird 2011) could not be accessed. The largest NZFS colonies on Banks Peninsula by pup production were Horseshoe Bay (485 ± 20 SE), Te Oka Bay (418 ± 1 SE) and Peraki Bay (312 ± 5 SE), all of which are on the peninsula's southern coastline.

4.5.1 Implications for pinniped responses during oil spills

The population and geographic expansion of any seal species in an area with a high oil spill risk classification (Navigatus Consulting 2015) is something that managers need to be aware of when planning efficient and successful responses to oil spills (Battershill et al. 2016).

During a spill, the welfare of charismatic megafauna like NZFS would receive significant interest from the public and the media (Paine et al. 1996), particularly as many pinniped species are the subjects of eco-tourism operations.

Preventing wildlife from becoming oiled should always be the primary objective in oil spill responses (Ziccardi et al. 2015; Hong et al. 2020), and is particularly important for fur seals. With other pinnipeds (e.g. phocids and sea lions), there may be greater opportunities for rescue and rehabilitation, as effective treatment would likely be less time intensive, due to these species' relative lack of fur (Ziccardi et al. 2015). Although, again, it is likely that only certain age classes of other pinnipeds could be rescued, due to the sizes that adults of some species attain. Additionally, some phocids, such as leopard seals, ribbon seals (*Histriophoca fasciata*) and bearded seals (*Erignathus barbatus*) do not form large terrestrial aggregations, reducing their likelihood of mass oiling. Contrastingly, the difficulties associated with successfully rescuing and rehabilitating fur seals once oiled (Mearns et al. 1999; Ziccardi et al. 2015; Chilvers 2017; Northwest Area Committee 2019), and the potential for mass oiling (Gales 1991; Mearns et al. 1999), mean that prevention, rather than rescue and rehabilitation, should be the primary objective. There are typically three ways to achieve this goal: (1) containing the oil as quickly as possible, (2) diverting oil away from habitats, (3) deterring or hazing wildlife from areas with spilled oil (IPIECA 2014; Chilvers and McClelland 2023; Chilvers 2024).

To assist with these three aims, projecting where spilled oil may travel is important to wildlife responses (Keramea et al. 2021). Currently, in New Zealand, tracking of spills using trajectory modelling, aerial surveys and shoreline observations are among the methods that can be used to follow slick movements (Maritime New Zealand 2021). When combined with current species abundance and distribution data, such as those provided in this chapter, these surveys and projections can allow responders to forecast how many animals may be impacted by a spill and deploy resources and personnel accordingly. The results from this study suggest that an oil spill projected to impact the south-western coast of Banks Peninsula (Figure 4.2) would likely affect the most NZFS, as opposed to the south-eastern or north-eastern coasts (Figures 4.3 and 4.4), as the southern-western coastline has more, larger colonies than either of the other two coastlines. Given the importance of prompt interventions during oil spills to protect wildlife (Ziccardi 2015), knowledge of where the densest aggregations of locally present species are likely to be found should speed up the effective deployment of personnel and resources to the locations where they are most needed (Grubesic et al. 2019).

Various methods exist for containing, diverting or removing spilled oil. These include physical barriers (booming) (Ghaly and Dave 2011), in situ oil burn offs (Fritt-Rasmussen et al. 2023), chemical dispersants to degrade oil droplets and assist natural weathering (Prince 2015), and bioremediation (Okeke et al. 2022). Each of these methodologies has strengths and weaknesses (Mullin and Champ 2003; Yang et al. 2009; Ghaly and Dave 2011; Dimitrakiev et al. 2020), including limitations based on local environmental conditions (Yang et al. 2009; Ghaly and Dave 2011), regulations and response practices. For example, in situ burning is often avoided close to coastlines due to concerns for human health and the risk of secondary fires (Ghaly and Dave 2011; Fritt-Rasmussen et al. 2023), and is not permitted at all within New Zealand waters.

On Banks Peninsula, several common remediation technologies are unlikely to be viable and/or desirable for protecting NZFS from oil spills. For example, dispersants can be toxic to marine taxa (Muncaster et al. 2016), and can cause loss of waterproofing in birds, replicating the impacts of oil itself (Whitmer et al. 2018). Additionally, booming is unlikely to work along much of the southern and south-eastern coastlines of Banks Peninsula, where the densest NZFS colonies are found (Figure 4.2 – 4.4) due to high wave action, which can wash oil over these barriers (Ghaly and Dave 2011). This reality is likely to be repeated in other fur seal habitats, as these species mainly inhabit rocky shores, which are typically high energy. Sites on Banks Peninsula where booms could be deployed include Haylocks Bay (Figure 4.3; No. 19), where NZFS pups are mostly at the head of this very narrow bay, away from swells, or Red Bay and Shell Bay (Figure 4.3; Nos. 26-27) which are less directly exposed. Even at these sites, however, booming viability would depend on ocean and weather conditions during a spill. Booms could also be used effectively to contain oil spilt within Lyttelton and Akaroa Harbours. The waters in these areas are often relatively calm and, at Lyttelton in particular, the port entry is narrow and could be effectively sealed.

While ports are particularly likely to experience oil spills (Battershill et al. 2016), the grounding of the *Austro Carina* (Figure 4.5), a 25-metre-long fishing vessel, on Banks Peninsula near Red Bay (Figure 4.1, No. 26), in September 2023, served as a reminder that oil spills can occur anywhere vessels travel. Despite carrying 10,000 litres of diesel and 400 litres of hydraulic oil (Radio New Zealand 2023a), and grounding near NZFS and spotted shag colonies, there were no reports of oiled wildlife following the *Austro Carina* incident (Environment Canterbury, pers, comm.), most likely because the grounding occurred at a time of year when fewer animals are typically present.

Figure 4.5 The grounded *Austro Carina*, near Red Bay. Photo taken on October 25th, 2023

Difficulties with deploying common oil containment and diversion equipment around Banks Peninsula mean that preventing or deterring NZFS from accessing impacted areas likely represent more effective mitigation options. Such approaches fall into two broad categories – pre-emptive capture (Chilvers and McClelland 2023) and deterrence/hazing (Chilvers 2024). With NZFS, both techniques would likely be required, given the differences in behaviour and habitat use across NZFS at different ontological stages (Crawley and Wilson 1976). For oil spill response planning, the most salient difference is between pre-weaned pups (hereafter, 'pups'), which are largely confined to their natal colony (Berkson and DeMaster 1985), and the remaining weaned individuals (hereafter, 'non-pups') which forage in the ocean and come ashore for rest and, at certain times of the year, for breeding and pupping (McNab and Crawley 1975; Crawley and Wilson 1976).

Pups typically remain in their natal colony while their mothers are away foraging (McNab and Crawley 1975), and, when still in their lanugo coats, cannot thermoregulate effectively meaning they cannot spend extended time in the water (Erdsack et al. 2013). Combined, these factors mean pups would be unlikely to vacate the coastline if oil was present, which could lead to mass oiling (Gales 1991; Mearns et al. 1999). In accessible colonies with low pup numbers, like Te Kaio (Figure 4.2; No. 8), Red Bay (Figure 4.3; No. 26) or Lavericks Bay (Figure 4.4; No. 33), pre-emptive capture of pups (and any non-pups that can be safely contained) may be viable (Chilvers and McClelland 2023). NZFS pups fast while their mothers are away foraging (Harcourt et al. 2002), and could be contained for a few days *in situ* without feeding at most times of the year, provided their health was monitored. Ideally, as many non-pups as safely possible would also be contained ashore, as individuals oiled at sea returning to the colony would likely oil the terrestrial habitat and animals they encounter while ashore. Pre-emptive capture could allow time for oil to be removed from coastal substrates (Wolfaardt et al. 2008) or from the ocean (Ghaly and Dave 2011). This strategy would, however, be logistically challenging and resource and personnel-intensive (Chilvers and McClelland 2023), meaning it could only be maintained for a few days. As such, the emphasis should remain on promptly containing oil movement offshore, and removing oil from the ocean surface through mechanisms such as booms and skimmers (Ghaly and Dave 2011) and bioremediation (Okeke et al. 2022).

During spills, non-pups risk oil exposure both on land and at sea. The only viable option for sub-adult and adult males is monitoring, as their size would limit safe capture and handling, and even attempting to rescue and rehabilitate smaller non-pup females would likely be difficult (Chilvers 2017). Therefore, the best option for preventing oiling of non-pups would likely involve attempting to haze or deter fur seals away from areas where oil was present. Hazing wildlife involves using negative stimuli to move them out of an area, while deterrence uses unpleasant or fearful stimuli to engender an escape or avoidance response to prevent wildlife from entering an area (Chilvers 2024). Such methods have been used to keep pinnipeds away from fishing operations (Lehtonen et al. 2022), prevent predation on endangered fish species (Tidwell et al. 2021) and deter them from potentially harmful underwater sound sources (Mikkelsen et al. 2017), with mixed results. For example, numbers of Steller sea lions (*Eumetopias jubatus*) preying on endangered salmon on the Columbia River were reduced during hazing involving seal bombs (underwater pyrotechnics), paintballs, rubber bullets and cracker shells, however, sea lion behaviour returned to normal once hazing ceased (Tidwell et al. 2021). Additionally, avoidance responses declined over time, suggesting habituation (Tidwell et al. 2021). Mixed success has also been reported with acoustic harassment devices (AHDs), which proved successful in a study of grey seals (*Halichoerus grypus*) (Vetemaa et al. 2021) but not in another involving harbour seals (*Phoca vitulina*) (Mikkelsen et al. 2017).

Non-pups could easily be hazed off oil-impacted shorelines as these individuals typically flee to the water when humans approach (pers. obs.). However, this may not be desirable if oil remained in the water around the colony, and, as such, should only be considered in the later stages of a response, when water-borne oil has been contained or removed, and the intention is to clean fouled shorelines (Mearns et al. 1999). More practicable would be to haze or deter NZFS away from oil slicks. This would likely be achieved through a combination of pinniped hazing and deterrence devices such as AHDs and pyrotechnics (Mikkelsen et al. 2017; Tidwell et al. 2021; Vetemaa et al. 2021). Such devices could also be deployed in the vicinity of the colonies identified in this study (Figures 4.2 – 4.4) to deter oiled non-pups from returning to unoiled colony habitats. However, whether it would be possible to deter highly philopatric NZFS (Bradshaw 1999) from returning to their own colonies is unknown, particularly if the individual was a mother returning to feed her pup. It is advised that trials of hazing or deterrence devices on NZFS should be performed prior to their deployment during an oil spill to assess their efficacy and evaluate potential impacts on other species. For example, there are concerns that seal bombs can cause temporary or permanent hearing loss in both pinnipeds and cetaceans (Simonis et al. 2020), the latter of which react to audio deterrence mechanisms at greater distances (Mikkelsen et al. 2017). The presence of Hector's dolphins around Banks Peninsula (Carome et al. 2022), as well as transitory cetaceans (Gibbs et al. 2018), means that the impacts of hazing and deterrence on other species must be considered.

Regardless of methodology, deterrence or hazing would likely need to be continuous for the duration of the risk to NZFS to be effective (Tidwell et al. 2021). Success would also depend on where the oil was relative to NZFS foraging grounds. At sea, NZFS are mostly either foraging or travelling to or between foraging sites (Page et al. 2005). Lactating female NZFS show fidelity both to their foraging sites and the approximate routes taken from their colonies (Baylis et al. 2012). Given this site fidelity, and the fact that it is difficult to discourage animals away from easily accessible and abundant resources (Simonis et al. 2020; Tidwell et al. 2021), it may be harder to deter or haze NZFS from spilled oil within their foraging grounds, compared to oil slicks between colonies and foraging grounds. Foraging ecology studies of Banks Peninsula's NZFS, like those conducted in Australia (Page et al. 2006; Hoskins et al. 2017), would complement the current study in identifying high-use habitats for local NZFS, and thus aid timely oil spill responses. Currently, it is assumed that Banks Peninsula's NZFS forage over the continental shelf and shelf edge (Allum and Maddigan 2012), but the precise locations of foraging grounds are unknown. Once foraging grounds were located it would be useful to know whether hazing and deterrence device efficacy differed between foraging and non-foraging sites, to promote efficient resource use during a spill.

While a pinniped response during a marine oil spill would be complex at Banks Peninsula, it would likely be less logistically difficult than if such an event occurred in other, less accessible parts of New Zealand, such as lower Fiordland, where a similar sized NZFS population (13,971 – 24,000) was recently surveyed (Chilvers 2021). Fiordland is another area popular with cruise tourism, and its extremely remote location, frequent poor weather conditions and challenging terrain (Egan et al. 2023) could impede timely wildlife responses. More challenging again would be a spill impacting one of New Zealand's subantarctic islands, which are also visited by cruise boats, and where some NZFS populations have not been monitored in ca. 30 – 40 years (Moore and Moffat 1990; Taylor 1992). Increases in global cruise tourism in recent decades, combined with trends towards cruise vessels visiting remote and/or ecologically important areas (Cerveny et al. 2020; Lau et al. 2023), as well as expansions of commercial shipping (Lau et al. 2023) make oil spills in such logistically challenging environments more likely (Chilvers et al. 2021). As such, it is important that both ecological data are collected for these areas, and that area-specific response plans are developed and in place. In remote regions, this planning should include optimum locations for housing response equipment, as well as safe and efficient access routes for response personnel. Additionally, comparatively accessible locations, such as parts of Banks Peninsula, as well as requiring their own response plans, offer the opportunity to trial response equipment, for example hazing apparatus, for use with populations of conspecifics in remote areas.

4.5.2 Changes in NZFS abundance and distribution

Twenty-five of the 41 NZFS colonies identified in this study had not been previously recorded. Both the distribution and abundance of NZFS have increased when compared with previous surveys on Banks Peninsula (Ryan et al. 1997; Bradshaw et al. 1999; Boren et al.

2006b), with the spread of colonies appearing to follow Roux' (1987) description of fur seal recolonisation patterns, and Bradshaw et al.'s (2000b) suggestion that new colonies are founded close to existing ones, in a density-induced spillover effect. This pattern has previously been suggested for Banks Peninsula (Boren 2005; Emami-Khoyi et al. 2016a), and appears to have continued. The two colonies for which previous comparable data existed have both grown, but only slightly. Te Oka Bay (Figure 4.2, No. 7) increased from ~300 pups in 2005 (Boren et al. 2006b) to 418 (± 1 SE) pups in 2024, while Horseshoe Bay (Figure 4.2, No. 13) grew from ~300 pups in 2005 (Boren et al. 2006b) to 485 (± 20 SE) pups in 2024. These relatively small increases relative to other NZFS colonies in New Zealand (Hall et al. 2024 (Chapter 2)), combined with the founding of new colonies, again suggests spillover (Bradshaw et al. 2000b), while the substantial geographic expansion of NZFS breeding on Banks Peninsula highlights the need for more regular population surveys to inform oil spill response plans. This regularity is particularly important in regions where pinnipeds are recolonising (Arnould et al. 2000; Shaughnessy and Goldsworthy 2015; Milano et al. 2020a), to ensure that plans' ecological bases still reflect reality. In areas where fur seal population monitoring has not occurred for some time, knowledge of the spillover pattern of breeding distribution could help inform the search and recovery phase of an oil spill wildlife response (Hunter et al. 2019). Suitable, but previously unused, habitat close to known colonies may have become occupied since monitoring last occurred (Bradshaw et al. 2000b; Hall et al. 2024 (Chapter 2)), and thus could be prioritised when searching for oiled animals. However, it is preferable that regular population monitoring is conducted to ensure that the ecological foundations of oiled wildlife response plans remain current for a given area (Battershill et al. 2016).

In 2023/24, more pups were produced on the southern coastline relative to the remainder of Banks Peninsula, and no breeding was recorded west of Squally Bay on the north coast

(Figure 4.4). There are also no NZFS colonies within Akaroa or Lyttelton Harbours. There are several potential explanations for this. Generally, NZFS are thought to be recolonising New Zealand from South to North (Dussex et al. 2016), meaning that initial recolonisation (Wilson 1981) and subsequent spillover likely favoured Banks Peninsula’s south coast. Additionally, a lack of preferred terrestrial habitat features (Ryan et al. 1997; Bradshaw et al. 1999) in some northern bays and Akaroa Harbour, and/or the amount of vessel traffic in Lyttelton and Akaroa Harbours (Boren et al. 2002; Cowling et al. 2014) may have deterred NZFS colony foundation. These absences are also important to understand for planning oil spill responses. Ports are among the places where oil spills are most likely to occur (Battershill et al. 2016), but the absence of NZFS colonies in the immediate vicinity of both Lyttelton and Akaroa Harbours (Figures 4.2 – 4.4) means that, initially, NZFS swimming in the ocean nearby are more likely to be impacted than individuals ashore. If oil cannot be contained within these ports, the emphasis should be on modelling the likely direction the oil slick will move, to understand which, if any colonies, could be impacted. This will aid in the timely deployment of resources and personnel to threatened locations. Simultaneously, hazing or deterrence should be employed from the water to prevent NZFS from swimming into the slick.

A three-yearly monitoring program is suggested for Banks Peninsula’s NZFS to enable timely detection of changes to population trends and allow managers to update regional oil response plans, which are also updated every three years (Environment Canterbury pers, comm). This timeframe would provide the benefits of more regular surveying, as described with a similar framework in Australia (McIntosh et al. 2018a) and would be consistent with the monitoring program currently in place at declining NZFS colonies on the West Coast of the South Island (Roberts and Neale 2016). Another benefit of regular monitoring is the potential to use the large, high-trophic level NZFS as sentinels for marine ecosystem change (Moore 2008; Bossart 2011), including for anthropogenic pollution (Brock et al. 2013; Donahoe et al. 2014;

Taylor et al. 2018; Fulham et al. 2018) and the impacts of climate change (Elorriaga-Verplancken et al. 2016).

4.6 Conclusion

This study confirms the presence of an expanding NZFS population in an area of New Zealand with a high risk of oil spills (Navigatus Consulting 2015). The NZFS population estimate of 13,147 – 17,675 calculated here is considerably higher than any previous estimate for Banks Peninsula's NZFS (Wilson 1981; M. Morrissey 2007, unpublished data; Baird 2011), and new colonies have been founded beyond those already recorded (Ryan et al. 1997; Boren 2005; Baird 2011). Recovering NZFS populations around New Zealand, and pinniped populations in other countries (Kirkwood et al. 2010; Milano et al. 2020b), means there are likely to be greater incidences of deleterious interactions between these species and human infrastructure and activities, including oil acquisition and transport facilities. As such, thorough and regularly reviewed plans need to be in place to inform oiled pinniped responses.

Given the potentially catastrophic consequences of oil spills for fur seals (Mearns et al. 1999; Ziccardi et al. 2015), preventing spilled oil from reaching land, and hazing or deterring pinnipeds away from slicks, should be prioritised by responders during a marine oil spill. To aid preparation, trials should be conducted to determine the efficacy of hazing and deterrence methods on NZFS, as no such studies have previously been conducted with this species. Additional areas of study that would be beneficial to oil spill responders, and in the general management of Banks Peninsula's NZFS, include regular future monitoring of their distribution and abundance, as well as foraging ecology studies to ascertain the locations of favoured feeding grounds for individuals within this population.

Chapter 5 Towards an abundance estimate for New Zealand fur seal in New Zealand

New Zealand fur seal pups at Cape Palliser, North Island

This chapter is a reformatted version of the following manuscript, currently accepted for publication by Aquatic Conservation: Marine and Freshwater Ecosystems

Hall, A. A., Chilvers, B. L. & Weir, J. S. (*in press*). Towards an abundance estimate for New Zealand fur seal in New Zealand

MASSEY UNIVERSITY
TE KUNENGA KI PŪREHUROA
UNIVERSITY OF NEW ZEALAND

GRADUATE RESEARCH SCHOOL

STATEMENT OF CONTRIBUTION
DOCTORATE WITH PUBLICATIONS/MANUSCRIPTS

We, the student and the student's main supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the student's contribution as indicated below in the Statement of Originality.

Student name:	Alasdair Hall
Name and title of main supervisor:	Prof. Louise Chilvers
In which chapter is the manuscript/published work?	5

Describe the contribution that the student and members of the supervisory team have made to the manuscript/published work:[1]

All authors to the study conception, methodology design and to data collection. Alasdair Hall and Louise Chilvers contributed to funding acquisition, data analysis and visualisation. The first draft of the manuscript was written by Alasdair Hall and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Please select one of the following three options:

(●) **The manuscript/published work is published or in press**

Please provide the full reference of the research output:

Hall, A. A., Chilvers, B. L. & Weir, J. S. (in press). Towards an abundance estimate for New Zealand fur seal in New Zealand. Aquatic Conservation: Marine and Freshwater Ecosystems.

( ) **The manuscript is currently under review for publication**

Please provide the name of the journal:

( ) **It is intended that the manuscript will be published, but it has not yet been submitted to a journal**

Student's signature:	Alasdair Hall — Digitally signed by Alasdair Hall Date: 2025.03.16 14:29:12 +13'00'	Main supervisor's signature:	Barbara Louise Chilvers — Digitally signed by Barbara Louise Chilvers DN: cn=Barbara Louise Chilvers, c=NZ, o=Massey University, ou=Wildbase, School of Veterinary Science, email=b.l.chilvers@massey.ac.nz Date: 2025.03.17 09:55:51 +13'00'

This form should be placed at the beginning of each relevant thesis chapter.

[1] Refer to the Massey University Publishing and Authorship guidelines (OneMassey for staff, Stream for students) and/ or Contributor Roles Taxonomy (CRediT) guidelines for guidance.

5.1 Abstract

A lack of population abundance and trajectory data is a conservation and management issue relevant to numerous pinniped species, many of which are exposed to a variety of threats. New Zealand fur seal (*Arctocephalus forsteri*; 'NZFS') populations in different parts of New Zealand have experienced both substantial increases and decreases to their abundance over the last 50 years, since the last nationwide census. Here, existing data and stage-structured matrix modelling were used to provide a contemporary nationwide estimate of NZFS abundance. Graphical depictions demonstrate the spatial inconsistencies in NZFS monitoring in New Zealand through time. A minimum population estimate of 131,338 – 168,269 NZFS was calculated by combining the most recently available pup production data from around New Zealand and using established multipliers. A second estimate of 181,646 – 239,473 NZFS was calculated using stage-structured matrix models to project contemporary abundance. Inconsistent NZFS population monitoring and sparse vital rate data for New Zealand's NZFS limited this study, and both population ranges are likely underestimates. However, they still represent substantial increases on the most cited nationwide abundance figure (100,000 NZFS). From these findings, we suggest that a regularised program of monitoring is adopted for New Zealand's NZFS, as has been achieved for similar species in other countries. This would both aid in the management of NZFS in the face of emerging risks, such as H5N1 avian influenza, and enable their use as a sentinel for the health of New Zealand's marine ecosystems.

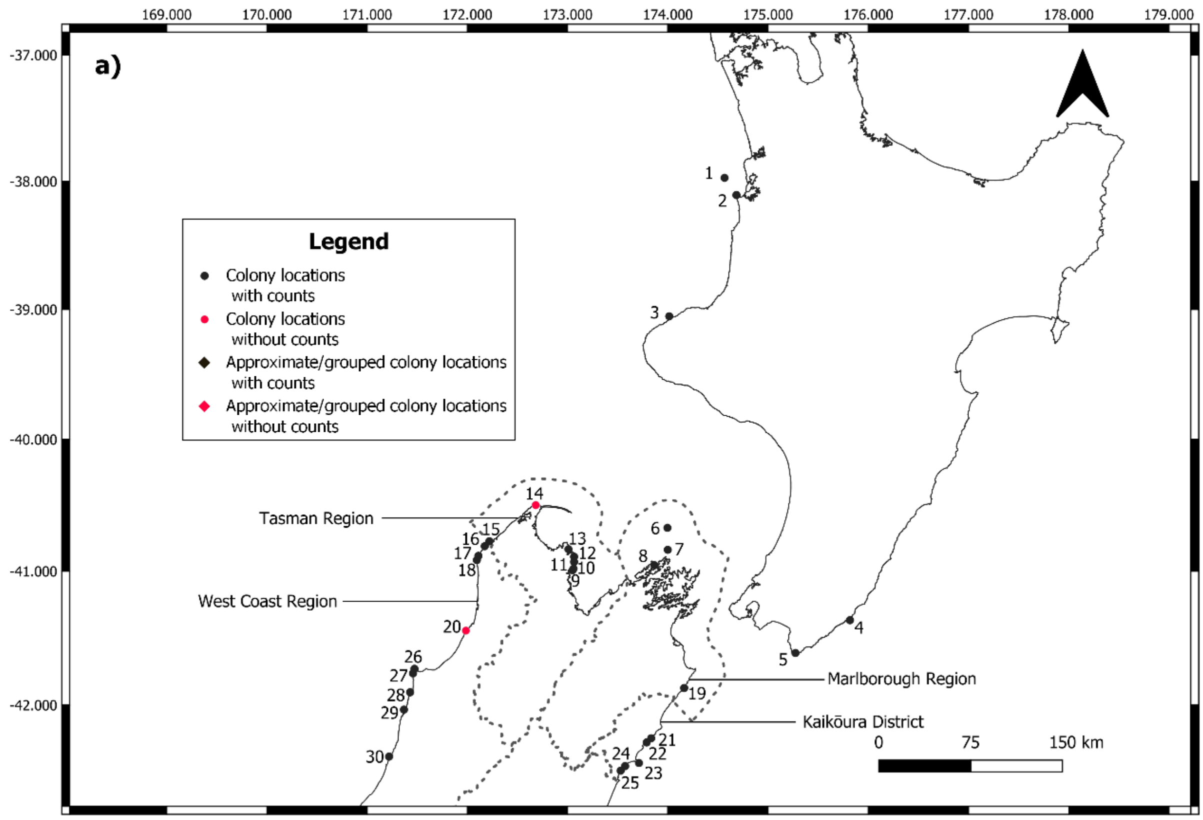
5.2 Introduction

Population abundance estimates are important to species management and conservation (Wittmer et al. 2010; Wege et al. 2016). Without contemporary abundance estimates, it can be difficult to prioritise species for conservation interventions (Bland et al. 2015) or identify emerging threats (Zipkin and DiRenzo 2022; Campagna et al. 2024). Outdated understandings of population size and trajectory can lead to risks being missed and interventions delayed, sometimes with devastating consequences (Voyles et al. 2014). Widely distributed, or wide-ranging, species often experience varying population trajectories (Roberts and Neale 2016), and differing threats through space (Lascelles et al. 2014; Wright et al. 2022). Macro understandings of abundance are required to recognise the impacts of disparate threats on a species, and because policies and agreements governing conservation action, and other relevant anthropogenic activities, are often set at national or supranational levels (Robinson et al. 2014).

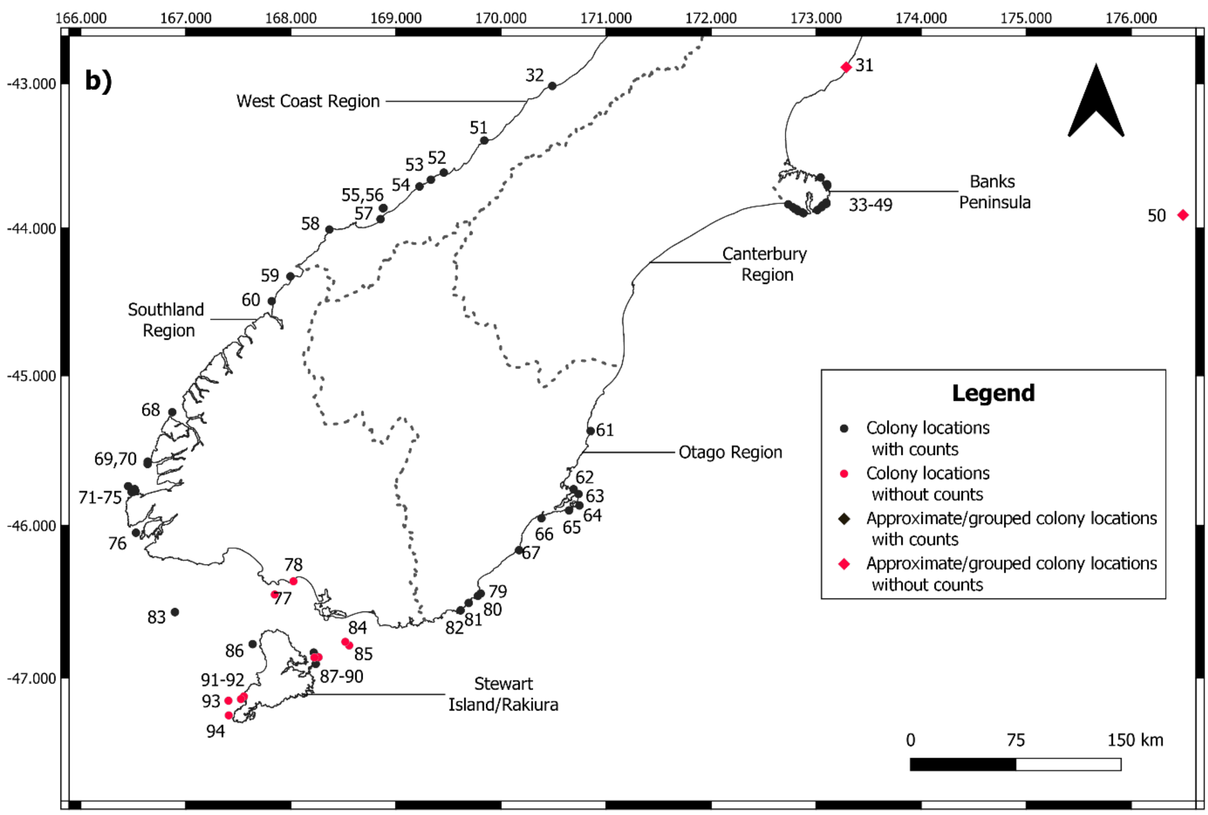
A lack of population abundance and trajectory data is common to many pinniped species around the world, making population level change hard to detect (Kovacs et al. 2012). Other than Otago (MacDiarmid and Lalas 2014) and three colonies on the West Coast of the South Island (WCSI) (Roberts and Neale 2016) (Figure 5.1), there is no longitudinal New Zealand fur seal/kekeno (*Arctocephalus forsteri*; 'NZFS') abundance data collection for colonies within New Zealand waters. Here, New Zealand waters refers to mainland New Zealand, plus all nearshore and offshore islands within its Exclusive Economic Zone. Most colonies/regions have only been surveyed sporadically, and, in extreme examples, some have gone unassessed for nearly 40 years (Taylor 1992). Additionally, differing survey methodologies (Taylor et al. 1995; Gooday et al. 2018; Chilvers 2021) make it challenging to compare results. For example, counts conducted from aeroplanes (Baker et al. 2009) have been shown to produce underestimates relative to ground counts (Stringell et al. 2014). Counts from boats (Taylor et al. 1995) are also typically less reliable than those conducted terrestrially (pers. obs.). There has also been little demographic parameter data collection for NZFS in New Zealand (Mattlin 1978b; Lalas and Harcourt 1995; Bradshaw 1999; Dickie and Dawson 2003; Boren 2005), although such data do exist for NZFS in Australia (Goldsworthy 1994; Mckenzie et al. 2005; Mckenzie 2006; Goldsworthy and Page 2007). The lack of NZFS vital rate data from New Zealand means that multipliers borrowed from elsewhere in the species' range (Goldsworthy and Page 2007), or from other Arctocephalus spp. (Taylor 1992) have been used to scale up NZFS pup production estimates into population estimates (Chilvers 2021; Hall et al. 2024 (Chapter 2)).The most commonly used multiplier (Goldsworthy & Page, 2007) derives from lifetables of NZFS from Australia, and allows estimation of an entire colony or population abundance from the numbers of pups produced. Pup production estimates are the most common way of assessing pinniped population abundance (Berkson and DeMaster 1985) (see Section 5.3), although there are drawbacks to such approaches, for example if a given population is not at a stable age/stage distribution when assessed, the abundance of non-pup cohorts cannot be reliably deduced from pup counts (Karamanlidis 2024).

There is currently no comprehensive estimate for NZFS abundance within New Zealand waters (hereafter, 'nationwide abundance'), as the species continues its post-exploitation recovery (Dussex et al. 2016). A figure of 100,000 is the most cited nationwide abundance estimate (Taylor et al. 1995; Wickens and York 1997; Harcourt 2001), and 200,000 has been suggested for the combined New Zealand-Australia NZFS population for over 20 years (Chilvers & Goldsworthy, 2015). These figures are based on the last attempted nationwide census, between 1971 – 1974, which estimated a population of 30,000 – 50,000 NZFS (Wilson 1981). However, this census was incomplete, and count reliability varied between sites (Wilson 1981). The uncertainty of the subsequent estimates is demonstrated by the fact that 200,000 NZFS has been suggested in 2015 for both New Zealand's population (L. Boren pers. obs, in Emami-Khoyi et al., 2018), and for the combined New Zealand – Australia population through cumulative regional estimates (Chilvers & Goldsworthy, 2015). Consequently, these figures can be best thought of as guesstimates. In 2014, an estimate of

117,400 individuals was suggested for Australia's NZFS abundance (Chilvers and Goldsworthy 2015) based on the most recently available data at the time. The abundance of NZFS in Australia has increased by approximately 60% in the last 20 years (Shaughnessy and Goldsworthy 2015). Recent studies, based on robust pup production estimates, have provided NZFS abundances for New Zealand regions that had not been assessed for some time, such as lower Fiordland (2021) (Chilvers 2021), Kaikōura (2023) (Hall et al. 2024 (Chapter 2)) and Banks Peninsula (2024) (Chapter 4). The locations of currently known NZFS colonies are shown in Figure 5.1, based on a literature review and discussions with NZFS researchers and local conservation managers. Of the limited number of recently assessed sites, nearly all are thought to be growing (Chilvers 2021; Hall et al. 2024 (Chapter 2)). However, three colonies on the WCSI (Cape Foulwind, Wekakura Point and Taumaka Island; Figure 5.1) are documented as in decline (Roberts and Neale 2016). Overall, the monitored colonies in New Zealand are unlikely to be representative of some areas that have gone unassessed for long periods – such as New Zealand's subantarctic islands, where environmental and ecological conditions are very different to the mainland.



a)
Legend
Colony locations with counts
Colony locations without counts
Approximate/grouped colony locations with counts
Approximate/grouped colony locations without counts
Tasman Region
West Coast Region
Marlborough Region
Kaikōura District
0 75 150 km



b)
West Coast Region
Banks Peninsula
Canterbury Region
Southland Region
Otago Region
Stewart Island/Rakiura
Legend
Colony locations with counts
Colony locations without counts
Approximate/grouped colony locations with counts
Approximate/grouped colony locations without counts
0 75 150 km

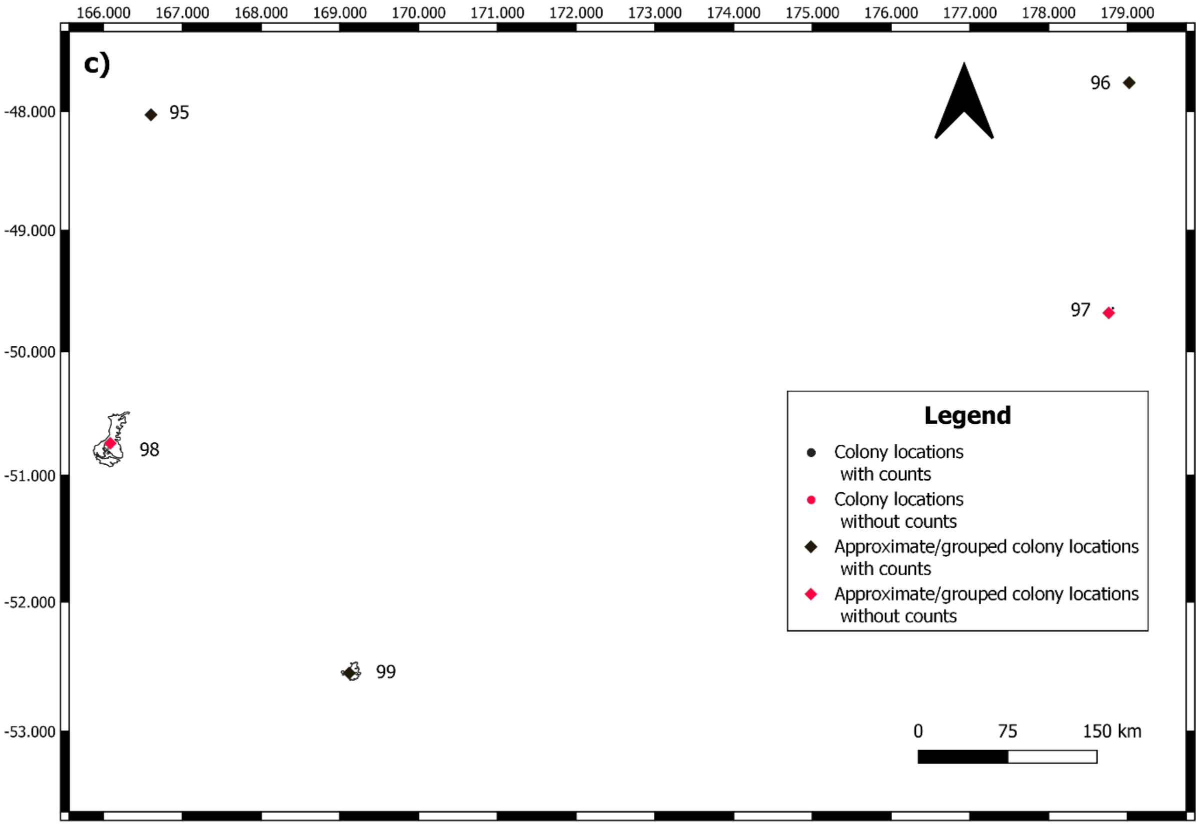


Figure 5.1 Known New Zealand fur seal colonies in New Zealand. 1.Gannet Island, 2. Albatross Point, 3. Sugarloaf Islands, 4. Honeycomb Rock, 5. Cape Palliser, 6. Stephens Island, 7. Trio Islands, 8. Admiralty Bay, 9. Fisherman Island, 10. Adele Island, 11. Pinnacle Island, 12. Tonga Island, 13. Separation Point – Shag Harbour, 14. Cape Farewell, 15. Kahurangi Point, 16. Otukoroiti Point, 17. Steep Point, 18. Wekakura Point, 19. Needles Point, 20. Kongahu Point, 21. Ōhau Point, 22. South Rakautara, 23. Kaikōura Peninsula, 24. Kaikōura South Coast, 25. Otumatu, 26. Black Reef, 27. Cape Foulwind, 28. Charleston, 29. Seal Island, 30. Point Elizabeth, 31. North Canterbury, 32. West of Pukutuaro Cliff, 33. Tokoroa Bay, 34. Te Oka Bay, 35. Robin Hood Bay, 36. Peraki Bay, 37. Horseshoe Bay, 38. Island Bay, 39. Whakamoa Bay, 40. Pōhatu Bay, 41. Reef Nook, 42. Sleepy Bay, 43. Otanerito Bay, 44. Shell Bay, 45. Goat Pt, 46. Paua Bay, 47. Ducksfoot Bay, 48. Pa Bay, 49. Long Lookout Point, 50. Chatham Islands, 51. Gillespie's Point, 52. Hanata Island, 53. Abbey Rocks, 54. Rock off Knight's Point, 55. Taumaka Island, 56. Popotai Island, 57. Jackson Bay, 58. Cascade Point, 59. Long Reef Martins Bay, 60. Yates Point, 61. Moeraki Peninsula, 62. Heyward Point, 63. Otago Peninsula – North, 64. Otago Peninsula – Centre, 65. Otago Peninsula – South, 66. Green Island, 67. Quoin Point, 68. Nee Island, 69. Breaksea Island, 70. Wairaki Island, 71. Five Finger Peninsula, 72. Anchor Island, 73. Many Islands, 74. Outer Islands, 75. Seal Island, 76. Chalky Island, 77. Rarotoka/Centre Island, 78. Rocks, Riverton, 79. Nugget Point, 80. Tucks Cove, 81. Penguin Bay, 82. Cosgrove Island, 83. Solander Island, 84. Ruapuke Island, 85. Breaksea Islands, 86. Whenhua Hou/Codfish Island, 87. Edwards Island, 88. Herekopare Island, 89. Bunkers Islets, 90. Bench Island, 91. Kundy Island, 92. Big Island, 93. Mokinui/Big Moggy Island, 94. Taukihepa /Big South Cape Island, 95. Snares Island, 96. Bounty Islands, 97. Antipodes Islands, 98. Auckland Islands, 99. Campbell Island. Note: NZFS breed in most bays and headlands on the southern and south-

eastern coasts of Banks Peninsula (33 – 49). To keep this as concise as possible, only colonies with 50+ pups (Chapter 4)) in this area are labelled here. Where colony locations are not precisely known, or multiple colonies have been grouped for an area, for example on the subantarctic islands, this has been made shown in the key

The current highly pathogenic avian influenza (HPAI) H5N1 pandemic, which has killed birds and mammals (Plaza et al. 2024), and been diagnosed in humans (Pulit-Penaloza et al. 2024), is among the most compelling contemporary reasons for monitoring New Zealand's NZFS abundance. HPAI is estimated to have caused mortality of over 17,400 southern elephant seals (*Mirounga leonina*) in Argentina in 2023 (Campagna et al. 2024) and correlates with unusual mortality events (UMEs) in harbour seals (*Phoca vitulina*) and grey seals (*Halichoerus grypus*) (Puryear et al. 2023), as well as South American sea lions (*Otaria flavescens*) (Ulloa et al. 2023). Given HPAI's detection within Antarctic and subantarctic regions (Bennison et al. 2024), and within *Arctocephalus* spp., the virus will likely reach New Zealand (Gartrell et al. 2024). Otariids can be effective indicators for the presence of pathogens (Donahoe et al. 2014), and sentinels for changes to the marine environment (Brock et al. 2013; Elorriaga-Verplancken et al. 2016; Taylor et al. 2018), which can aid effective species and wider ecosystem management.

Substantial difficulties exist when assessing wildlife abundances at large geographic scales, particularly for long-lived species (Baker et al. 2016; Christie et al. 2016; Torney et al. 2019), with modelling often offering the best approaches (McDonald et al. 2014; Ureña-Aranda et al. 2015; Fleming et al. 2022). For vertebrate species which experience synchronised birth pulses, such as fur seals, age-structured models (Leslie 1945), or their stage structured derivatives (Lefovitch 1965), are the foundation of most population modelling approaches (Schaub and Kéry 2022), and have previously been used to project population abundances in various pinniped species (Meyer et al. 2015; Davis 2022; Forcada et al. 2023). Age and stage-structured models rely on the ability to observe changes in the factors which influence population trajectories (vital rate data) such as survival rates (the probability of an individual surviving from time *t*-1 to time *t*), and fecundity rates (the rate of offspring production) as an organism progresses through its life (Schaub and Kéry 2022). Where such data are lacking for every age group in a population but can be approximated for stages of an organism's existence (e.g. egg, tadpole, juvenile frog, adult frog (Govindarajulu et al. 2005)) stage-structured models can be used (Gales and Fletcher 1999). When applying such models to project population abundances, effort should be made to account for factors which may impact vital rates over time, such as population density (Murphy et al. 2023) and both demographic (Eacker et al. 2017; Manlik et al. 2022; Murphy et al. 2023) and environmental stochasticity (Horswill et al. 2022; Manlik et al. 2022). However, allowing for such factors in population models requires additional data on how, when and to what extent they impact population trajectories, which is currently missing for NZFS in New Zealand.

As such, several approaches were taken in this chapter to provide an updated nationwide abundance estimate for New Zealand's NZFS, by combining recent surveys and stage-structured population modelling. From these methods, a preferred model was selected based on performance, and its results tested against non-modelled data to assess its realiablity. In addition to seeking to provide an updated nationwide abundance estimate for NZFS, this chapter aims to highlight longstanding inconsistencies in NZFS monitoring methodologies and make suggestions on how future population monitoring could be improved.

5.3 Materials and methods

Three approaches were undertaken to estimate NZFS's nationwide abundance in New Zealand: a minimum population abundance estimate (5.3.1), a density-independent projection model (5.3.2.1) and a density-dependent projection model (5.3.2.2).

NZFS abundance estimates are typically extrapolated from pup production estimates (Boren 2005; Roberts and Neale 2016; Chilvers 2021), because pups younger than ca. 3-4 months are easily identifiable and confined to their natal colonies. Thus, theoretically, this entire cohort can be counted, unlike older individuals, of which an unknown proportion may be at sea during surveys (Berkson and DeMaster 1985). Direct counts (Taylor et al. 1995; Taylor 1996; Gooday et al. 2018) and mark-recapture (Boren et al. 2006b; Chilvers 2021) are the two most common ways of estimating NZFS pup production. Direct counts are suitable at smaller colonies (Chilvers 2021) but produce underestimates compared with mark-recapture at larger sites (Watson et al. 2009), although the former are quicker, cheaper, less invasive and require fewer personnel.

5.3.1 Minimum nationwide abundance estimate

A minimum nationwide NZFS abundance estimate was calculated using the most recent abundance data from colonies/regions for which such information was available (Table 5.2) without any projection forward. Where only pup counts were provided, these were scaled up to population estimates using Goldsworthy and Page's multiplier (2007) (Chilvers 2021). Where provided in the original study, population estimates were used directly. For populations where abundance ranges were provided, the lower and upper bounds were separately summed with the single estimates from the remaining sites to produce a total range.

Pup production and population estimates were collected through a literature review and by seeking unpublished data from researchers. Figure 5.2 provides a summary of the geographic distribution of NZFS abundance research in New Zealand over time. A more detailed breakdown is given in Appendix 4.

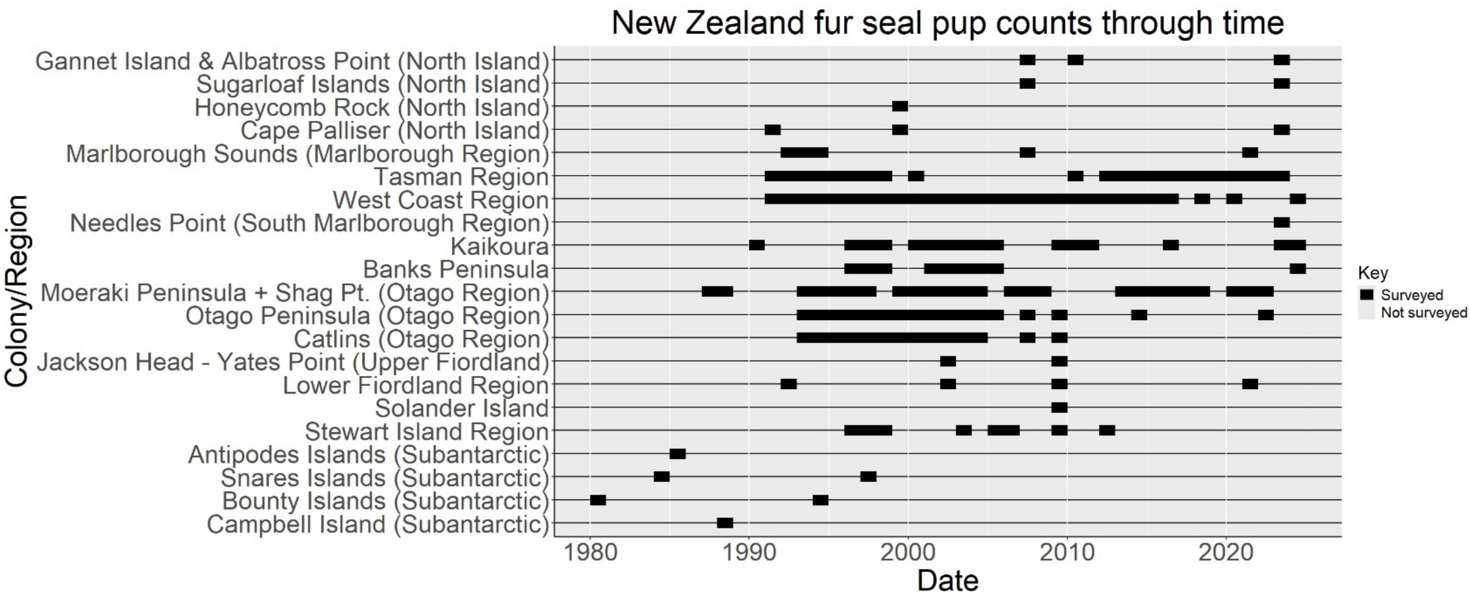


Figure 5.2 Summary of the geographic distribution of New Zealand fur seal abundance studies through time in New Zealand since Wilson (1981). Years marked as "surveyed" mean that an abundance study took place at that region in that year. Mostly, these studies did not cover the entire region. Where isolated colonies occur, these are included individually – this comprises all North Island colonies, and Needles Point

5.3.2 Population projection modelling

Two modelling approaches were applied, using stage-structured matrix models (Lefkovitch 1965; Caswell 2001). Both defined three stages of NZFS ontogeny: "pups" (0-1 years old), "juveniles" (1-4 years old), "adults" (4+). The cut-off between juvenile and adult is arbitrary, but age 4 was used in many of the studies from which vital rate data were derived (Appendix 5). The models can be described as:

$$\boldsymbol{n}_{t+1} = \mathbf{A}\mathbf{n}_t$$

With $\mathbf{n}_t$ and $\mathbf{n}_{t+1}$ representing the number of individuals in each stage at time intervals t and t+1 respectively, and **A** representing the population matrix:

$$\mathbf{A} = \begin{pmatrix} 0 & 0 & \omega \\ \varphi_1 & \varphi_2\sigma & 0 \\ 0 & \varphi_2(1-\sigma) & \varphi_3 \end{pmatrix}$$

Table 5.1 Parameter description for the matrix population models used to estimate New Zealand fur seal abundance

Symbol	**Parameter**
φ_1	Pup survival
φ_2	Juvenile survival
φ_3	Adult survival
ω	Adult fecundity
σ	Juvenile transition

Vital rate data are required for matrix models, however these are highly limited for New Zealand's NZFS (Mattlin 1978b; Dickie and Dawson 2003). As such, additional vital rate data were applied from other fur seal species (Appendix 5).

Additional to survival rates for each stage, and fecundity rates for adults, a transition rate was required from the juvenile – adult stages. This refers to the rate of transition between defined stages, typically due to maturation, and here is calculated as:

$$1 - \frac{1}{x_j - x_i}$$

Where x_i is the first age class in stage i (juveniles; 1) and x_j is the first age class in stage i + 1 (adults; 4) (Fujiwara and Diaz-Lopez 2017). In this instance, giving 0.66.

To be forward projected, colonies/regions had to:

- Have gone unassessed for the last five years (before the year of writing; 2024).
- Have more than 100 pups at their last count.
- Not represent counts listed as "guesstimates" in the literature.
- Not represent populations which are thought to have stabilised (no longer growing).

These criteria were selected for the following reasons. Due to the limitations imposed by the lack of data, particularly regarding the effects of density dependence on NZFS population growth, and the frequency with which NZFS colonies/regions experience variation in vital rates over time (see Sections 5.3.2.2 and 5.5.3), it was decided that estimates from colonies/regions assessed within the last five years would likely be more accurate than projected population estimates.

Colonies with fewer than 100 pups at their last count were excluded due to the instability of small populations, resulting from their greater susceptibility to the impacts of stochastic factors (Galimberti et al. 2001; Davison et al. 2019). Counts listed as "guesstimates" in the literature (e.g. some counts in Baird 2011) were excluded due to a lack of information on how these figures were reached. Finally, colonies/regions with populations that are thought to have stabilised were not included, as it seemed inappropriate to project their numbers forward, when evidence suggests they are no longer growing. Three populations with previous pup counts are thought to have stabilised: Snares Island, Otago/the Catlins and Solander Island. For Snares Island and Otago/the Catlins, determinations of the populations having stabilised were based on the conclusions of the researchers who studied them (Carey 1998; MacDiarmid and Lalas 2014). For Solander Island, this determination was based upon the results of two population assessments, made 36 years apart, which indicated no substantial change in local NZFS abundance (Wilson 1981; Baker et al. 2009).

Additional to the colonies excluded by the above rules, Popotai Island, where 118 pups were counted from a plane in 2009 (Baker et al. 2009) was also excluded. Popotai is approximately 500 metres from Taumaka, where regular monitoring (Department of Conservation 2023) has shown consistent declines in pup production. While we do not know definitively that pup production on Popotai Island is also declining, it seems likely, given the potential role that fisheries bycatch and competition could be playing in pup production declines on Taumaka and other proximal colonies (Roberts and Neale 2016). As such, it was deemed inappropriate to project Popotai Island's population forward.

5.3.2.1 Density-independent projection model

Roux (1987) suggests that density dependent factors may eventually slow colony growth for recolonising *Arctocephalus* spp., and Bradshaw et al. (2000b) show that new NZFS colonies are founded near existing ones, potentially representing a compromise between the species' philopatry and resource or space depletion at the older colonies. However, there are insufficient data demonstrating when density-dependent factors may start to impact NZFS population growth, and such effects would likely vary by location. As such, a density-independent modelling approach was trialled using the *pop.projection* function within the *popbio* package (Stubben and Milligan 2007) in R Studio (R Core Team 2022). This method produced such improbably large results (see Section 5.5.2) that a full description of the methods and results is not included.

5.3.2.2 Density-dependent projection model

Due to the evidence for the impacts of density dependence on *Arctocephalus* spp. colony growth (Roux 1987; Bradshaw et al., 2000b), a density-dependent modelling approach was implemented using the *stoch.proj* (Stubben 2020) function in the *popbio* package (Stubben and Milligan 2007) in R Studio.

Minimum, average and maximum values were selected for each required vital rate, from the data in Appendix 5. The juvenile-adult transition rate was calculated at 0.66, as outlined above (Fujiwara and Diaz-Lopez 2017). Five matrices were then created. The “Low”, “Average” and “High” matrices contained, respectively, the lowest, the average and the highest values for each vital rate. The fourth matrix (High Survival, Low Fecundity) contained average rates for all inputs, except adult survival rates were high and adult fecundity rates were low. The final matrix (High Fecundity, Low Survival) was the inverse regarding adult fecundity and survival. These final two matrices were included due to evidence of reproductive costs on female Antarctic fur seal (*Arctocephalus gazella*) survival (Boyd et al. 1995).

Starting abundances are required for each ontogenetic cohort to initiate the projections. These were calculated for the juveniles and adults in each projected region/colony by using popbio’s *stable.stage* function (Stubben and Milligan 2007). The stable stage refers to the distribution of individuals within a population across the defined stage cohorts (e.g. pup, juvenile, adult), once that population has reached its asymptotic growth rate (Schaub & Kerry, 2022). Stable stages were first calculated for each of the five matrices, and then a single stable stage calculated by averaging those results for each respective ontogenetic cohort. As the starting abundances for pups were already known, the proportions provided by the stable stage could be used to provide the same for juveniles and adults.

The *stoch.proj* function permits inclusion of simple density-dependence by providing a maximum number of individuals (*nmax*) that the projections cannot exceed (Stubben 2020). Given the lack of data, for individual colonies (e.g. Stephens Island), *nmax* was set as the largest pup production estimate hitherto obtained for a single NZFS colony in New Zealand (2,471 at Ōhau Point in 2015; (Gooday 2016)) multiplied by 4.76 (Goldsworthy and Page 2007), giving an *nmax* of 11,761. For multiple colonies across regions, e.g. Stewart Island, the Otago/Catlins area population estimate (28,800) was used. This is the largest regional NZFS population estimate recorded in New Zealand (MacDiarmid and Lalas 2014). The impacts of selecting these *nmax* values are reported for each projection (Table 5.3), and a discussion of the limitations of this approach is provided in Section 5.5.3.

stoch.proj also enables the prescription of a weighted probability vector for matrix selection at each iteration. Little data exist on the frequency with which NZFS experience variation in vital rates through time, making it difficult to select values for this vector. Expert opinion indicates that the 'average' vital rate matrix should occur more frequently than any other matrix. As such, this was given a higher probability weighting, and the remaining four matrices were given equal weightings. Nineteen possible integer combinations exist whereby one value is largest, and the remaining four equal, with all five summing to 100. To demonstrate the impacts of varying the probability weightings on the final population estimates, three results are presented for each projection (Tables 5.3 and 5.4). These represent the "high difference" (Average = 0.96, other = 0.01), "median difference" (Average = 0.6, other = 0.1) and "low difference" (Average = 0.24, other = 0.19) relative differences in the probability values ascribed to the Average versus the other four (Low; High; High Survival, Low Fecundity; High Fecundity, Low Survival) matrices.

Each projection ran for 1,000 iterations, and the respective projection intervals equalled the numbers of years elapsed between the last local NZFS survey and 2024.

5.3.2.3 Model testing

The results of the density-dependent model were tested using data collected from Moeraki Peninsula in the Otago/Catlins region by C. Lalas (pers. comm.), and results from Te Oka Bay on Banks Peninsula (Boren et al. 20006b; Chapter 4).

The Moeraki data set contains pup count estimates for 23 of the years between 1987 – 2021, with pups first recorded in 1988 (C. Lalas, unpublished data). This dataset was projected from 2001, when 96 pups were estimated, until the most recent survey in 2021, when 782 pups were estimated (plausible range 606 – 964) (C. Lalas, unpublished data).

The Te Oka Bay record is less complete, with data available in 1997, 2001 – 05 and 2024. The value of 84 pups from 2003 (Boren et al. 2006b) was used to initiate the projection, and model results were compared with those from 2024 (Chapter 4), where 418 ± 1 SE pups were estimated at Te Oka Bay in 2024.

The initiation years for Moeraki (2001) and Te Oka Bay (2003) were selected as the pup counts in these respective years were the closest to the minimum 100-pup rule described in Section 5.3.2. The methodology was as described in Section 5.3.2, with the “high difference”, “median difference” and “low difference” weighting variations tested on the data, and *nmax* for both was set at 11,761.

5.4 Results

5.4.1 Minimum nationwide abundance estimate

A total minimum nationwide pup count of 27,798 – 34,538 was calculated, based on the most recently available, unprojected pup counts listed in Table 5.2. From this, a minimum nationwide abundance of 131,338 – 168,269 NZFS was calculated, derived from the sum of the most recent pup counts, multiplied by 4.76 (Goldsworthy and Page 2007), or population totals provided in the original studies.

Table 5.2 Most recent New Zealand fur seal pup counts and derived minimum population estimates

Colony/region	Year of most recent pup production estimate	Count method	Last pup production estimate	Source	Minimum population estimate
Gannet Island & Albatross Point	2023	Boat direct counts	15	C. Hansen (pers. comm)	71
Sugarloaf Islands	2023	Boat direct counts	25	Chaddy's Charters (pers. comm)	119
Honeycomb Rock	1999	Direct count	10	(Baird 2011)	48
Cape Palliser	2023	Direct count	600	L. Boren, pers. comm.	2,856
Stephens Island	1994	Direct count	276 [A]	(Taylor et al. 1995)	1,314
Trio Islands	2007	Boat Direct counts	50	(Baird 2011)	238
Admiralty Bay	2021	Direct counts	100	L. Boren, pers. comm.	476
Tasman Region (Separation Point to Onetahuti, Tonga Island, Adele Island, Pinnacle Island and Fisherman Island)	2023	Direct counts	101	L. Boren, pers. comm.	481
Kahurangi/Otukoroiti	2009	Aerial direct count	63	Baker et al. 2009)	300
Steep Point	2009	Aerial direct count	25	Baker et al. 2009)	119
Wekakura Point	2023	Mark-recapture	144	(Department of Conservation 2023)	685

Black Reef	2010	Direct Counts	200 [A]	(H. Best, *unpublished data*)	952
Cape Foulwind	2023	Mark-recapture	93	(Department of Conservation 2023)	443
Charleston	2010	Direct Counts	476 [A]	(H. Best, *unpublished data*)	2,266
Seal Island	2009	Aerial direct counts	6	Baker et al. 2009)	29
Point Elizabeth	2003	Direct Count	10	(Baird 2011)	48
Needles Point	2023	Direct count	36	L. Boren (pers. comm.)	171
Kaikōura (Ōhau Point – Otumatu)	2023	Mark recapture and direct counts	4,400 – 5,781	(Hall et al. 2024)	20,944 – 27,518[B]
Banks Peninsula (all colonies)	2024	Mark recapture and direct counts	2,762 – 3,713	(Hall et al. 2024, *in review*)	13,147 – 17,675[B]
West of Pukutuaro Cliff	2009	Aerial direct counts	16	Baker et al. 2009)	76
Gillespie's Point	2009	Aerial direct counts	3	Baker et al. 2009)	14
Hanata Island	2009	Direct counts	66	(Baker et al. 2009)	314
Abbey Rocks	2009	Aerial Direct counts	12	Baker et al. 2009)	57
Rock off Knights Point	2009	Aerial direct counts	13	Baker et al. 2009)	62

Taumaka Island	2023	Mark-recapture	776	(Department of Conservation 2023)	3,694
Popotai Island[C]	2009	Aerial direct counts	118	Baker et al. 2009)	NA
Jackson Head	2009	Aerial direct counts	21	(Baker et al. 2009)	100
Cascade Point	2009	Aerial direct counts	1,466[A]	(Baker et al. 2009)	6,978
Long Reef, Martins Bay	2009	Aerial direct counts	163[A]	(Baker et al. 2009)	776
Yates Point	2009	Aerial direct counts	1,228[A]	Baker et al. 2009)	5,845
Lower Fiordland (Doubtful/ Pateā, Dusky and Breaksea Sounds, and Chalky Inlet)	2021	Mark recapture and direct counts	2,935 – 5,042	(Chilvers 2021)	13,971 – 24,000[B]
Otago and the Catlins (Oamaru to Slope Point)	2009	Direct counts	2,993 – 5,294	(MacDiarmid and Lalas 2014)	13,000 – 28,800[B]
Solander Island	2009	Aerial direct counts	1,107	(Baker et al. 2009)	5,269
Rakiura/Stewart Island (and surrounding islands)	2003	Mark recapture and direct counts	2,694[A]	(Watson et al. 2015)	12,823
Snares Islands	1997	Direct counts	188	(Carey 1998)	895
Bounty Islands	1994	Direct counts	4,380[A]	(Taylor 1996)	20,849
Antipodes Islands	1985	Direct counts	7	(Taylor 1992)	33

Campbell Islands	1988	Direct counts	126[A]	(Moore and Moffat 1990)	600
Minimum estimate totals			**27,798 - 34,538**		**131,338 – 168,219**

[A] Indicates sites that met the criteria to be projected to 2024. [B] Indicates population estimates provided in the original study. [C] The 2009 estimate for Popotai was not used in any calculations due to declining pup numbers on nearby Taumaka Island but is included here for completeness.

5.4.2 Density-dependent estimate

When summed, the sites that did not meet the criteria to be projected (Table 5.2) produced an unmodelled population range of 77,375 – 125,044. The results of the projected sites/colonies are presented below (Table 5.3), combined with number of times the *nmax* ceiling was reached in the 1,000 iterations.

Using the results from the "high difference" scenario, which appear likely to be the most accurate (as shown in Section 5.4.3), the projected populations of the modelled colonies/regions totalled 104,271 – 114,429 (Table 5.3). When combined with the unmodelled colonies, a nationwide abundance range of 181,646 – 239,473 NZFS was calculated.

Table 5.3 Estimated contemporary New Zealand fur seal population abundances for modelled colonies/regions

Colony/ region	Latest recorded pup count (year)	Projected 2024 pup abundance (mean ± SD)	Projected 2024 population abundance (mean ± SD)	No. times nmax reached (/1000)	Projected 2024 pup abundance (mean ± SD)	Projected 2024 population abundance (mean ± SD)	No. times nmax reached (/1000)	Projected 2024 pup abundance (mean ± SD)	Projected 2024 population abundance (mean ± SD)	No. times *nmax* reached (/1000)	Maximum range
		"High difference"	"High difference"	"High difference"	"Median difference	"Median difference"	"Median difference"	"Low difference"	"Low difference"	"Low difference"	
Stephens Island[A]	276 (1994)[1]	2,427 – 3,134	9,414 – 11,946	190	1,319 – 2,905	5,214 – 10,865	166	736 – 2,516	3,031 – 9,290	77	3,031 – 11,946
Black Reef[A]	200 (2010)[2]	525 – 654	2,038 – 2,488	0	329 – 730	1,325 – 2,700	0	239 – 752	981 – 2,738	0	981 – 2,738
Charleston[A]	476 (2010) [2]	1,260 – 1,578	4,908 – 5983	0	817 – 1,761	3,242 – 6,521	0	591 – 1,748	2,433 – 6,368	2	2,433 – 6,521
Cascade Point[A]	1,466 (2009)[3]	2,913 – 3,201	11,523 – 11,936	903	2,379 – 3,457	9,646 – 12,381	578	1,919 – 3,423	7,968 – 12,183	357	7,968 – 12,381
Long Reef, Martins Bay[A]	163 (2009)[3]	460 – 583	1,787 – 2,205	0	298 – 677	1,203 – 2,464	0	206 – 644	842 -2,353	0	842 – 2,464
Yates Point[A]	1,228 (2009)[3]	2,905 – 3,210	11,432 – 11,994	908	2,220 – 3,364	8,896 – 12,317	479	1,740 – 3,361	7,133 – 11,955	307	7,133 – 12,317
Stewart Island Region[B]	2,694 (2003)[4]	7,158 – 7,806	28,143 – 29,275	883	5,910 – 8,345	23,847 – 30,406	558	4,583 – 8,316	18,888 – 29,847	335	18,888 – 30,406
Bounty Islands[B]	4,380 (1994)[5]	7,178 – 7,810	28, 229 – 29,224	847	6,259 – 8,408	26,075 – 29,806	611	5,471 – 8,575	28,842 – 30,051	446	22,842 – 30,051
Campbell Island[B]	126 (1988)[6]	1,758 – 2,458	6,797 – 9,378	0	713 – 2,487	2,806 – 9,333	0	813 – 8,228	813 – 8,228	1	813 – 9,378
Projected site totals		26,584 – 30,434	104,271 – 114,429		20,244 – 32,134	82,256 – 116,792		16,298 – 37,563	64,932 – 113,013		64,932 – 118,202
Unmodeled sites total											77,375 – 125,044
Grand total*											181,646 – 239,473

[1] (Taylor et al. 1995), [2] (Baird 2011), [3] (Baker et al. 2009), [4] (Watson et al. 2015), [5] (Taylor 1996), [6] (Moore and Moffat 1990). "High difference", "Median difference" and "Low difference" refer to the probability weightings assigned to the matrices, explained in Section 5.3.2.3.* This figure combines the unmodelled total with the modelled results from the "high difference" scenario, which appears likely to be the most accurate (Section 5.4.3).[A] Used *nmax* of 11,761, [B] used *nmax* of 28,800.

5.4.3 Model testing results

The results of the model testing from Moeraki Peninsula and Te Oka Bay are presented in Table 5.4. The figures in this table refer to pup counts, rather than population estimates, in contrast to Table 5.3, as pup counts are comparable with the observed results from the original studies. Across the "High", "Median" and "Low" probability weightings, the population *nmax* of 11,761 was hit just once out of the 1,000 iterations on the "High" scenario for Moeraki, and not at all in the others.

Table 5.4 Results of the model testing

Colony/ region	Pup count in year initiated	True pup count in final year of study	Projected pup abundance (mean ± SD)			Maximum range (based on means ± SD)
			"High difference"	"Median difference"	"Low difference"	
Moeraki Peninsula	96 (2001)[1]	782 (2021)	457 ± 62	396 ± 172	343 ± 204	111 – 717
Te Oka Bay	84 (2003)[2]	418 ± 1[3] (2024)	432 ± 59	370 ± 158	327 ± 210	117 – 537

[1] (C. Lalas, unpublished data), [2] (Boren et al. 2006b) and [3] (Chapter 4). "High difference", "Median difference" and "Low difference" refer to the probability weightings assigned to the matrices, explained in Section 5.3.2.2.

5.5 Discussion

This study is the first empirical effort to update our understanding of New Zealand's nationwide NZFS abundance since Wilson (1981). A minimum nationwide NZFS abundance estimate of 131,338 – 168,269 was calculated using the latest unmodelled population figures and pup production estimates, combined with Goldsworthy and Page's (2007) population multiplier. A second estimate of 181,646 – 239,473 NZFS was produced using stage-structured matrix models that projected previous pup production estimates forward to 2024. Combining the modelled abundance figure with Australia's NZFS population estimate (117,400; (Chilvers and Goldsworthy 2015), gives a range-wide abundance estimate for NZFS of 299,046 – 356,873.

Uncertainties around some existing pup production figures, and the methodologies' limitations, discussed below, highlight the need for more regular and methodologically consistent NZFS monitoring within New Zealand. Such monitoring would benefit both NZFS management and wider understandings of New Zealand's marine ecosystems (Kirkman et al. 2007; Elorriaga-Verplancken et al. 2016).

5.5.1 Minimum nationwide abundance estimate

The minimum nationwide abundance estimate (131,338 – 168,269) is an increase on the most cited nationwide abundance estimate of 100,000 (Taylor et al. 1995; Harcourt 2001), but smaller than the 200,000 NZFS figure cited in Emami-Khoyi et al. (2018). It is worth remembering, however, that much of the data input for these earlier population estimates are guesstimates. While the new minimum nationwide abundance estimate presented here provides a baseline, it is unlikely to accurately reflect New Zealand's current NZFS abundance, as many pup production estimates are outdated – for example, by 39 years for the Antipodes Islands' population (Taylor 1992). Pup production can change dramatically over such periods - for example, between 1990 – 2023, pup production at Ōhau Point (Figure 5.1b), increased from three pups (M. Morrissey, *unpublished data*) to 2,401 (± 99) (Hall et al. 2024). While it is possible that some of the colonies/regions projected forward in this study may have, in fact, declined since their most recent assessments, as has occurred on the WCSI (Roberts and Neale 2016), the preponderance of the evidence from NZFS population studies across New Zealand and Australia over several decades suggests population growth, or stabilisation following a period of growth, are the most likely trajectories for the colonies/regions modelled here (MacDiarmid and Lalas 2014; Shaughnessy and Goldsworthy 2015; Chilvers 2021; Hall et al. 2024 (Chapter 2)).

Given the survey gaps at some sites and known growth rates from other NZFS colonies (Lalas and Harcourt 1995; Lalas and Murphy 1998; Lalas and Bradshaw 2001; Boren et al. 2006b) (Table 5.2), the minimum nationwide abundance figure is likely an underestimate. Additionally, some NZFS colonies have never been formally assessed (Baird 2011), and others may not have been discovered. The former was true of several colonies in Fiordland prior to 2021 (Chilvers 2021), Kaikōura prior to 2023 (Hall et al. 2024 (Chapter Two)), and Banks Peninsula prior to 2024 (Chapter 4). Additionally, known sites where no reliable pup count data exist, such as the Auckland Islands, were excluded.

The use of uncalibrated direct counts (Table 5.2) is another reason for suspecting that the minimum abundance figure is an underestimate (Watson et al. 2009). As no appropriate calibration indices exist for most parts of New Zealand, this could not be avoided. Additionally, some sites (Table 5.2) were assessed through photographs taken from aircraft (Baker et al. 2009). Aerial counts can be an unreliable way to survey seal populations (Phillips and Mathews 2007), and may produce underestimates even relative to terrestrial direct counts (Stringell et al. 2014). Surveying from boats also tends to produce underestimates relative to terrestrial direct counts (pers. obs.) As no other data exist for these locations, these counts were included.

5.5.2 The density-independent projection model

The results from the density-independent projections were also unsatisfactory. For example, a pup production estimate of 1,222,605 was projected for the Bounty Islands in 2024. This seems impossible, given that, at the last assessment (4,380 pups in 1994), 50% of the available breeding habitat was already occupied (Taylor 1996).

Such overestimates emphasise the importance of including density-dependence in NZFS population projections. Like other species (Noad et al. 2019; Tanasovici et al. 2020), NZFS populations can grow exponentially (Boren et al. 2006b), sometimes for extended periods (Shaughnessy and Goldsworthy 2015). However, this is not indefinite, and growth rates eventually slow (Roux 1987; Carey 1998; MacDiarmid and Lalas 2014). Thus, projections ignoring density-dependence are likely to overestimate NZFS abundance.

5.5.3 The density-dependent projection model

When combined with the summed estimates from the unmodelled colonies, the density-dependent model produced a nationwide abundance estimate of 181,646 – 239,473. No recent nationwide abundance estimates are available for comparison, meaning its validity must be assessed in other ways.

The result of the sensitivity testing using C. Lalas' (*unpublished*) Moeraki Peninsula data found that the model projections substantially underestimated the true number of pups born in 2021 (Table 5.4). However, the results from the test using Te Oka Bay were closer to reality (Table 5.4). For both test sites, the "high difference" probability weighting results, whereby the Average scenario was given a 0.96 probability, and the remaining scenarios were each given a 0.01 probability, were the closest to reality. Calculating average annual pup production rates at the two test colonies helps explain why the Te Oka projection was relatively accurate, while Moeraki Peninsula was not. The Moeraki Peninsula projection ran over 20 years (2001 – 2021), during which the actual average annual exponential rate of increase in pup numbers (calculated as per Lalas and Murphy (1998) was 0.1. Over the 21 years (2003 – 2024) covered in the Te Oka test, the actual average annual exponential rate of increase in pup numbers was 0.08. The model, thus, failed to replicate the faster average growth rate at Moeraki compared to Te Oka. As such, if the colonies/regions projected in Table 5.3 experienced similar, lengthy periods of substantial year-on-year growth, these results could also be underestimates. Notably, MacDiarmid and Lalas (2014) state that the Otago/Catlins region, where Moeraki Peninsula is situated (Figure 5.1b) experienced particularly rapid NZFS recolonisation. As such, this growth rate may not have been replicated elsewhere in New Zealand. Local factors, such as food availability, likely influence NZFS population trajectories, meaning trends through space are not uniform (Boren et al. 2006b). In addition, Roux (1987) showed that fur seal colonies typically take some time to enter a phase of rapid growth, thus if a colony takes longer to arrive at that phase, its overall growth will be slower. A factor that further complicates direct comparisons between colony growth rates is the concept of spillover breeding, whereby density-dependent factors at established breeding sites lead to new colonies being founded nearby (Bradshaw et al. 2000b). Therefore, pup production increases at Te Oka Bay could have been higher than the 2003 – 2024 comparison suggests, but this breeding may have spilled over into neighbouring bays, and thus not been assigned to Te Oka Bay's 2024 total (Chapter 4). Indeed, this seems likely, given that Te Oka Bay, in the years following the 2003 count of 84 pups, grew rapidly to 210 (2004) and then 301 (2005) (Boren et al. 2006b), with the next estimate being 418 ± 1 (SE)in 2024, when several colonies in proximal bays were recorded for the first time (Chapter 4).

Immigration could also engender varying NZFS population growth rates around New Zealand. NZFS recolonisation likely spread northwards from refugia colonies on the South Island's south-western coast (Dussex et al. 2016), with no North Island breeding recorded until 1991 (Dix 1993). As founders are more likely to establish new breeding sites close to existing ones (Bradshaw et al. 2000b), immigration likely followed the same staggered northwards expansion. Therefore, colonies in southern areas of mainland New Zealand are likely to have consistently received immigrants at a faster rate, and thus to have grown faster than colonies further north. Indeed, much of the substantial annual population increases recorded in Otago colonies, such as Moeraki Peninsula (Figure 5.1b), has been attributed to immigration (Lalas and Bradshaw 2001; MacDiarmid and Lalas 2014), perhaps partly explaining the difference in the model test results.

Historic sealing records (Ling 1990), and notes made during previous assessments, provide other ways to analyse some of the model results. For example, the upper bound of the Bounty Islands pup production estimate (8,575) seems reasonable, given that ca. 50% of the available breeding habitat was occupied when the last estimate was made (4,380 in 1994) (Taylor 1996). In the intervening years, NZFS may have colonised the remaining habitat, with pup production approximately doubling. Additionally, between 1807 – 1809, ca. 53,500 NZFS skins were taken from the Bounty Islands (Ling 1990). As competition from fisheries and changed environmental conditions may mean that the Bounty Islands can no longer support pre-exploitation NZFS population levels (Kuhn et al. 2014), a total projected population estimate of 28,229 – 29,224 ("high difference" scenario result) is not improbable.

Overall, the modelled projections should be treated with caution, due to the significant underestimation of Moeraki Peninsula's 2021 pup production. However, the model test results from Te Oka Bay, together with information available from other modelled locations, such as the Bounty Islands, suggests other projections may be realistic. If sensitivity tests were conducted on other colonies, it is likely that there would have been substantial diversity in the reliability of their respective results. This, again, is largely a result of the dearth of region-specific vital rate data for most of New Zealand's NZFS colonies.

Another limitation of the projection model is setting *nmax*. This argument provides simple density-dependence (Stubben 2020), and has been used previously in this capacity (Crain et al. 2019). Ideally, *nmax* would differ by site based on knowledge of local factors impacting NZFS population growth, including available breeding habitat (Boren et al. 2006b), predation (Boveng et al. 1998), food availability and bycatch mortality (Roberts and Neale 2016; Pavanato et al. 2023a), past disruptions (Hall et al., 2024 (Chapter 2)), and anthropogenic impacts on the marine environment (Brock et al. 2013). However, very little such data exist for New Zealand's NZFS, and certainly not sufficient to apply a different, justifiable *nmax* to each projection. Thus, it was deemed best to apply two consistent *nmax* figures, based on knowledge of the respective sites. The impacts of *nmax* were greatest on the larger colonies/regions, and particularly prevalent on the "high difference" runs, where the Average vital rate scenario had a probability of 0.96. For both Cascade Point and Yates Point, *nmax* (11,762) was reached in over 90% of the "high difference" projection runs, and *nmax* (28,800) was reached in nearly 85% of the "high difference" runs for Bounty Islands, and over 88% of "high difference" runs for Stewart Island (Table 5.3). This stands to reason, given that these colonies/regions' populations were already substantial at the time of their respective most recent assessments and so should experience the limitations of density-dependence more frequently in the simulations (Roux 1987). However, it is not ideal that *nmax* had such a substantial impact on these results, since the selection of the two *nmax* values had limited support, and had the *nmax* ceiling been set higher the results that were constrained by *nmax* (Table 5.3) would have also been greater. That said, there is some justification for the selection of these figures. Single colonies used the 11,761-population figure, based on Gooday's (2016) study at Ōhau Point near Kaikōura. Ōhau Point experienced particularly rapid pup production increases, growing at ca. 32% per annum between 2002 – 2005, and pups there likely have a buffer against low food supply periods, thanks to proximity to the highly productive Kaikōura Canyon (Boren et al. 2006b). This may mean that Ōhau Point has some immunity to growth-restraining resource limitation that might, at times, limit pup production increases elsewhere in New Zealand (Boren et al. 2006b). Importantly, NZFS abundance at Ōhau Point also appears to have largely escaped lasting negative impacts from the 2016 Kaikōura earthquake (Hall et al., 2024 (Chapter 2)). At the time of their study, Bradshaw et al. (2000a) found that NZFS pups on the East Coast of the South Island, where both Ōhau Point and the Otago/Catlins Region are located (Figure 5.1b), were in generally better condition than those on the West. Of the nine colonies projected forward to 2024, five are on the West Coast, including the two (Cascade Point and Yates Point) where *nmax* was reached most frequently. Given the links between pup survival and condition in fur seals (Majluf 1992), and the possibility that the Kaikōura Canyon has promoted particularly rapid, and sustained growth rate at Ōhau Point (Boren et al. 2006b), it is possible that the Cascade Point and Yates Point colonies have not surpassed Ōhau Point's population, thus rendering 11,761 a reasonable *nmax* to apply as an upper limit. The other two regions where *nmax* was reached in most of the "high difference" runs (Stewart Island and the Bounty Islands) had already surpassed 11,761 (Table 5.2), meaning this could not be used as a population *nmax*. The 28,800 figure used as the regional *nmax* was derived from Lalas and MacDiarmid's (2014) study of the Otago/Catlins region (Figure 5.1b). NZFS recolonisation of this area was particularly rapid (MacDiarmid and Lalas 2014), and this region is also on the South Island's East Coast, where pups appear to be in better condition (Bradshaw et al. 2000a) and thus, potentially, have better survival rates than in other areas (Majluf 1992). As such, it was the best option for *nmax* for the remaining projected sites. The need for a ceiling for sites such as the Bounty Islands was demonstrated by the impossible pup production estimate of 1,222,605 projected for this region in 2024 by the density independent approach.

Assigning probability weightings to the matrix selections was another difficult element of designing the methodology, given the lack of data. The main difference in the results generated by the “high difference", “median difference” and “low difference” scenarios was in the standard deviation around the means. Consistently, the standard deviation was least for the “high difference” scenario, and greater for the other two. This makes sense because, in the “high difference” runs, the average scenario was being selected with a probability of 0.96, whereas the other two (“median difference” and “low difference”) had greater opportunity to vary. By presenting three sets of results from each projected colony/region, the uncertainty of the estimations is demonstrated.

Finally, stage-structured models inherently risk oversimplifying study species’ ontogeny. This is likely here, as vital rates can vary substantially within the three stages prescribed. For example, prime breeding years for adult female NZFS are thought to be ages 8 – 13, sandwiched between ages of lower fecundity, potentially due to, initially, inexperience and, latterly, senescence (McKenzie et al. 2007). While this variation was captured in the adult fecundity data (Appendix 5), separate stages were not created, as there were insufficient corresponding survival data. Similarly, the juvenile age category was not further divided, despite evidence from other *Arctocephalus* spp. that post-weaners suffer higher mortality rates than older juveniles (Beauplet et al. 2005). The vital rates used came from studies of NZFS and other fur seal species (Appendix 5). Sharing vital rate data between taxonomically or ecologically comparable species is not unusual where such information is lacking (Kindsvater et al. 2018; Fleming et al. 2022), however it is not ideal, as these inputs can clearly impact matrix projections. That said, similarities in vital rates between different fur seal species across large geographic areas have been noted (Wickens and York 1997), and this relative homogeneity is evident in the vital rates selected for this study (Appendix 5), lending support to this approach.

5.5.4 Management implications

Understanding contemporary NZFS abundance has several important management implications. NZFS can act as marine ecosystem sentinels, due to their conspicuousness and elevated trophic position (Moore 2008; Bossart 2011). The usefulness of pinnipeds as sentinels has been demonstrated with threats including climate change (Elorriaga-Verplancken et al. 2016), pollutants (Taylor et al. 2018) and zoonotic diseases (Donahoe et al. 2014).

NZFS species management would also benefit from updated contemporary abundance estimates. As animals making regular use of two environments, NZFS face a complex range of threats, both anthropogenic and natural. In New Zealand, these include very high rates of bycatch (Pavanato et al. 2023a), entanglement (Boren et al. 2006a), human infrastructure (Hall et al. 2023 (Chapter 3)), illegal killings (Department of Conservation 2022), fisheries competition (Roberts and Neale 2016), changes to the marine environment (Elorriaga-Verplancken et al. 2016) and oil spills (Mearns et al. 1999). Regular abundance estimates promote understandings of the cumulative risks of such threats on the species and thus enable appropriate responses (Soykan et al. 2008; Kobayashi et al. 2014).

Regular NZFS population monitoring would also benefit understandings of climate change impacts on New Zealand's coastal species and ecosystems. NZFS risk habitat loss due to increases to global sea levels (Kovacs et al. 2012), which could lead to habitat inundation, as predicted for northern elephant seals (*Mirounga angustirostris*) in California (Funayama et al.

2013). Rising sea levels will likely act synergistically with storm surges to inundate larger quantities of fur seal habitat more regularly (McLean et al. 2018), and, as some NZFS inhabit space-limited reefs and islets (pers. obs.), and other sites have been spatially squeezed by anthropogenic infrastructure (Hall et al., 2024 (Chapter 2)), sea level rise may force site abandonment.

5.5.5 Future monitoring of NZFS in New Zealand

The largely opportunistic monitoring of New Zealand's NZFS populations since Wilson (1981), has engendered lasting uncertainty over the species' abundance. Notably, regions at both extremes of NZFS' New Zealand range have received less attention than central areas such as the WCSI (Roberts and Neale 2016) and Otago/Catlins (MacDiarmid and Lalas 2014) (Figure 5.1). Estimates from New Zealand's subantarctic islands are between 27 years (Snares Islands; (Carey 1998) and 39 years old (Antipodes Islands; (Taylor 1992)), and as previous estimates from some of these islands would make up a significant proportion of the total New Zealand NZFS population estimate (Table 5.2), they should be prioritised for future surveys.

Similarly, the Stewart Island region was last assessed 12 years ago, when only one island was evaluated (Watson et al. 2015). Although Chilvers (2021) assessed several colonies within lower Fiordland, much of this region has never previously been surveyed. Fiordland colonies likely provided refugia from which recolonisation began (Dussex et al. 2016), meaning that unknown, and potentially large, colonies possibly exist there today.

A regularised program of NZFS monitoring is strongly advised for New Zealand. The benefits of such approaches have been demonstrated on the WCSI (Roberts and Neale 2016), and with Australian fur seal, where switching from ad hoc monitoring to a co-ordinated approach meant trends could be more reliably detected (McIntosh et al., 2018a). Such monitoring would also enable detection and study of unusual mortality events, as noted in Kaikōura in 2023/24 which saw over 1,000 NZFS die, as well as aborted foetuses and emaciated live animals (J. Weir, unpublished data). The improvements to Australian fur seal monitoring (McIntosh et al. 2018a) involved a range wide census of the species. While, ideally, this would be replicated for NZFS in New Zealand, practically, this is unlikely. With many endangered species, some of which are the subject of resource, personnel and time-intensive conservation programs (Manno and Young 2023; Digby et al. 2023; Agnew 2024), currently, New Zealand is unlikely to be able to support a nationwide NZFS monitoring program. This is exacerbated by the extreme difficulty of accessing NZFS colonies in remote locations, such as those on some of New Zealand's subantarctic islands.

As such, the next-best, and most realistic, approach to future NZFS monitoring in New Zealand should comprise regular assessments of several baseline sites, using consistent methodologies, to enable trend detection. Given the value of long-term data sets for population level studies (Clutton-Brock and Sheldon 2010), it is suggested that these baseline sites should include Kaikōura, the Otago/Catlins and the three colonies on the WCSI already monitored regularly (Taumaka Island, Cape Foulwind and Wekakura Point). Kaikōura, Otago/Catlins and the three WCSI colonies currently represent the most complete records of NZFS colonies in New Zealand (Figure 5.2), and the WCSI is also a priority due its declining pup production rate (Roberts and Neale 2016). Additionally, Cape Palliser should be included among the regularly monitored sites, as it is the only substantial NZFS colony on New Zealand's North Island and is proximal to the busy Cook Strait fishing and shipping channel. Given that the records for many of these sites include counts around the time of their respective origins, not only would ongoing assessment enable future trend detection, it may also reveal insights into the impacts of density-dependence as the colonies age (Bradshaw et al. 2000b). As WCSI colony monitoring currently occurs every 2 – 3 years (Department of

Conservation 2023), and because large-scale longitudinal surveys can be expensive and resource intensive, the same timeframe is suggested as a minimum for the monitoring at these other baseline sites. Consistent survey methodologies are important and will likely comprise direct counts and mark-recapture (Chilvers 2021). Importantly, at least one site per region should be assessed by both mark-recapture and direct counts to enable calibration index calculation (Watson et al. 2009). It is also recommended that updated abundance estimates are made for New Zealand's subantarctic islands, given the risk that these sites act as a conduit to mainland New Zealand for HPAI H5N1 (Gartrell, Jolly & Hunter 2024). The difficulties and expense involved in accessing these sites mean they are unlikely to be monitored regularly. At the same time as baseline site monitoring occurs, it is suggested that vital rate data are collected from the northernmost (Cape Palliser) and southernmost (Otago/Catlins) of the regularly assessed sites. These data are key to constructing lifetables for New Zealand's NZFS, which would benefit future population modelling. Here, Goldsworthy and Page's (2007) multiplier was used to convert pup production counts into population estimates. However, this multiplier derives from a study in South Australia (Goldsworthy and Page 2007) making it suboptimal for use in New Zealand, as vital rates can vary spatially (Raithel et al. 2007).

5.6 Conclusion

This study highlights the current difficulties of estimating New Zealand's nationwide NZFS abundance, largely due to *ad hoc* population monitoring in most regions since Wilson (1981), large gaps in the record, and a lack of data on important parameters such as vital rates and the impacts of density-dependence on population dynamics. The minimum nationwide population abundance of 131,338 – 168,269 NZFS calculated here is likely a significant underestimate, based on knowledge from recent regional surveys (MacDiarmid and Lalas

2014; Chilvers 2021; Hall et al., 2024 (Chapter 4). The second estimate of 181,646 – 239,473 NZFS, produced using stage-structured matrix projection models, may also be an underestimate, however projections for some regions, such as the Bounty Islands appear to be reasonable.

A regular program of NZFS monitoring is strongly suggested in New Zealand to benefit the management NZFS and enable their use as a sentinel for the health of New Zealand's marine environments (Elorriaga-Verplancken et al. 2016).

Chapter 6. General Discussion and Conclusion

This thesis aims to improve knowledge of New Zealand fur seal (*Arctocephalus forsteri*; 'NZFS') population distribution and abundance, both at a national scale and in two ecologically important regions of New Zealand (Banks et al. 2002; De Leo et al. 2010), and to assess two specific anthropogenic threats (vehicle collisions and oil spills) to the species. This chapter summarises the results presented in earlier chapters, explores the ecological and management implications of the findings, and suggests avenues for future research.

6.1 Summary of findings

Chapter Two assessed the impacts of the 2016 Kaikōura earthquake on the local NZFS population. Mark-recapture and direct count surveys revealed substantial changes to the breeding distribution and abundance of Kaikōura's NZFS, relative to the most recent pre-earthquake study (Gooday 2016), and several colonies were documented for the first time. A regional population estimate of 21,560 – 28,327 NZFS was calculated. From these findings, updates to the boundaries of the Ōhau Point New Zealand Fur Seal Sanctuary were suggested to better reflect the current distribution and abundance of local NZFS breeding, as were the implementation of additional terrestrial sanctuary type protections at other sites along the Kaikōura coastline where increased NZFS breeding was recorded.

Chapter Three explored the relationship between Kaikōura's NZFS and State Highway 1 (SH1) using data collected by Waka Kotahi/New Zealand Transport Authority (NZTA) on the locations of live or dead NZFS ('NZFS incidents') on the road. A substantial increase in annual average NZFS incidents was detected relative to a previous study (Boren et al. 2008). Hotspots for NZFS incidents on SH1 were located using Kernel Density Estimation Plus (KDE+) (Bíl et al. 2013, 2016, 2019), and changes in hotspot location through time were identified using Poisson modelling. Environmental, temporal and spatial factors were examined to gain a better understanding of their impacts on the timing and location of NZFS incidents. Significant relationships were found between several environmental variables and NZFS incidents, and autumn experienced significantly more incidents than any other season. The growth of the local NZFS population likely drove the noted increase in incidents, while changes to roading infrastructure likely explained the variation in hotspot location through time. The presence of NZFS breeding was the best predictor of where NZFS incidents were likely to occur. New road barriers have been implemented along sections of SH1 based on the findings from this study (Appendix 3), as discussed below.

Chapter 4 provided the first update to the distribution and abundance of NZFS on Banks Peninsula in ca. 20 years. This population is at risk of oil spills due to proximity to two ports, and knowledge of population parameters is important for oil spill response planning. Through mark-recapture and direct counts, a regional population estimate of 13,147 – 17,675 NZFS was calculated. NZFS breeding now occurs in most bays and many headlands along Banks Peninsula's South and Southeastern coastlines, representing an expansion from previous studies. Consideration was given as to how NZFS could be protected during an oil spill, given the lack of facilities for cleaning and rehabilitating oiled pinnipeds in New Zealand. Preventing oil from reaching colonies and hazing or deterring animals away from oil slicks should be high priority response actions for NZFS during oil spills.

Chapter 5 represents the first empirical attempt in ca. 50 years to update New Zealand's nationwide NZFS abundance estimate. By using the most recent available abundance data, and a population multiplier (Goldsworthy and Page 2007), a minimum nationwide abundance estimate of 131,338 – 168,269 NZFS was calculated. By using stage-structured projection models (Lefkovitch 1965; Caswell 2001), a second estimate of 181,646 – 239,473 NZFS was calculated. There is a high degree of uncertainty in both estimates, due to the age of some of the available colony count data, and the lack of vital rate data for New Zealand's NZFS. This study highlighted the need for more consistent monitoring of NZFS in New Zealand, which would benefit both NZFS management and allow managers to use NZFS as proxies for the health of New Zealand's marine ecosystems.

6.2 Ecological implications

6.2.1 NZFS distribution and abundance

The distribution and abundance studies in Kaikōura (Chapter 2) and Banks Peninsula (Chapter 4) highlighted several common trends. First, the NZFS population abundance and distribution in both areas had increased significantly since the most recent respective studies (Boren et al. 2006b; Baird 2011; Allum and Maddigan 2012; Gooday 2016), conforming with results from most NZFS studies in both New Zealand and Australia over the past 30 years (Dix 1993; Boren et al. 2006b; MacDiarmid and Lalas 2014; Shaughnessy and Goldsworthy 2015; Chilvers 2021). This suggests that NZFS are continuing their recovery from historic sealing (Dix 1993; Bouma et al. 2008; Dussex et al. 2016; Emami-Khoyi et al. 2018, Chilvers 2021), a trend also noted in other countries where pinnipeds experienced exploitation (Arnould et al. 2000; Shaughnessy and Goldsworthy 2015; Milano et al. 2020a,b). Notably, an unusual mortality event (UME) was recorded in Kaikōura in the year following the abundance study (Chapter Two), with the number of pups produced in the 2023/24 breeding season at Ōhau Point dropping to 1,182 (J. Weir, *unpublished data*) from 2,401 (± 99) in 2022/23 (Chapter 2). Post-mortem investigations, including histology, found no evidence of disease. Rather, the cause of death was deemed to be starvation or nutritional stress in all cases examined (J. Weir, pers. comm.). No signs of high pup mortality were noted in Banks Peninsula in the 2023/24 breeding season (Chapter 4), although the fact that no abundance studies had occurred in Banks Peninsula in ca. 20 years prior to Chapter 4 (Boren et al.

2006b) meant that direct year-on-year comparisons were not possible for any Banks Peninsula colonies. The Kaikōura NZFS UME highlights the drawbacks of single-year studies when assessing population sizes (Karanth et al. 2006), as such analyses only provide snapshots, and thus cannot be used to infer trends. As has been demonstrated on the WCSI, NZFS colonies that had been growing or stable can suddenly experience sharp declines in pup production (Roberts and Neale 2016), trends which can only be empirically demonstrated through multi-year studies.

The Kaikōura (Chapter 2) and Banks Peninsula (Chapter 4) studies also provided the first distribution and abundance records of previously undescribed sites within both regions. This supports the theory that NZFS breeding follows a spillover pattern, whereby new colonies are founded close to existing ones, due to density-dependent limits on growth at the original sites (Bradshaw et al. 2000b). For example, pup production at Te Oka Bay, on Banks Peninsula, had only increased slightly in ca. 20 years (Boren et al. 2006b), but new colonies had become established in neighbouring bays and headlands, likely representing spillover. Moreover, the existence of substantial yet previously unrecorded breeding colonies in both regions, highlights the *ad hoc* approach to NZFS monitoring in New Zealand since Wilson (1981). An estimated 1,074 – 1,090 pups were born at Kaikōura's South Coast colony, ca. 25 minutes' drive from Kaikōura, in the 2022/23 breeding season (Chapter 2), strongly suggesting that this colony has existed for some time without being assessed. Similarly, numerous, previously unassessed colonies were recorded on Banks Peninsula (Chapter 4) despite this area's relative accessibility. Again, it is impossible to know how long these previously unrecorded colonies had been in existence prior to the study in Chapter 4. Thus, in addition to known unrecorded NZFS colonies (Baird 2011), it is likely there are other, yet undiscovered colonies elsewhere in New Zealand.

Chapter Two provides the first assessment of the impacts of a significant natural disaster, the 2016 Kaikōura earthquake, on a pinniped population. While estimated pup production at Ōhau Point, the most noticeably affected colony, was almost identical to the last pre-earthquake study (Gooday 2016), breeding is now spread over ca. 4x the length of coastline. This is likely a response to the narrowing of the colony habitat due to State Highway 1 (SH1) being rebuilt closer to the ocean following damage during the earthquake. That NZFS bred at Ōhau Point immediately post-earthquake (Gooday and Goldstien 2018) and, until the 2023/24 UME (J. Weir, *unpublished data*), had apparently not experienced substantial population declines, demonstrates the species' capacity to cope with significant disruptions. NZFS' high degree of philopatry (Stirling 1971; Bradshaw 1999), and the excellent foraging habitat offered by the Kaikōura Canyon (De Leo et al. 2010), likely explains the continued use of Ōhau Point, despite a reduction in the terrestrial habitat quality (Gooday and Goldstien 2018). This observation adds to the body of evidence for philopatric species making potentially maladaptive decisions to remain in degraded habitats (Lai et al. 2007; Dudley et al. 2022). As such, while NZFS abundance at Ōhau Point was apparently not severely impacted by the 2016 earthquake, unlike other local wildlife populations (Cuthbert 2019), there could still be long-term impacts resulting from the event. For example, sections of this colony's habitat are now much narrower than they were previously and abut an 8-10-metre-high sea wall that is impassable to NZFS. This renders these areas more vulnerable to inundation resulting from predicted increasing storm surges and rising sea levels (McLean et al. 2018) which may lead to further redistribution of the breeding population in the future. The philopatry demonstrated by NZFS could also prove problematic in the event of an oil spill, as it may put populations at greater risk of mass-oiling, and thus high mortality (Riffaut et al. 2005), compared to species that disperse more widely.

The 2016 Kaikōura earthquake demonstrated the importance of having 'before' data to compare with 'after' a sudden and catastrophic event. Gooday's (2016) study was fortuitously timed to provide a snapshot of the Ōhau Point NZFS colony in the year prior to the earthquake, meaning inferences could be drawn from the results presented in Chapter Two, despite several years having elapsed since the event. Similarly, the impacts of human-induced disasters, such as oil spills, on wildlife cannot be properly understood without pre-event ecological baseline data in place (Battersill et al. 2016). While Chapter 4 provides such data for NZFS in Banks Peninsula, equivalent information is not being collected for NZFS in much of New Zealand, potentially hampering efforts to plan pinniped responses, and reducing the ability to fully understand an event's overall effects.

6.2.2 Interactions with anthropogenic infrastructure

The most recognised direct threat to pinnipeds from humans comes from fisheries interactions, in the form of bycatch and entanglement (Kovacs et al. 2012). This thesis focussed on two anthropogenic threats which are less commonly researched with regards to pinnipeds – human infrastructure and oil spills. While the latter has received some consideration in the context of the effects of pollution on pinnipeds (Kovacs et al. 2012), and where oil spills have had catastrophic impacts on pinnipeds (Gales 1991; Mearns et al. 1999), there has been little research into how to protect pinnipeds during such events, which was the focus of Chapter Four. The Kaikōura road study (Chapter 3) is the first time that a comprehensive study seeking to understand the relationship between a pinniped and a road has been attempted, although previous studies have noted the existence of this threat and provided some quantitative descriptions of its impacts on pinniped colonies (Hatfield and Rathbun 1999; Boren et al. 2008).

The findings of this thesis indicate that the ongoing recovery of NZFS will involve constantly evolving relationships between the species and anthropogenic infrastructure. For example, there is now a large population of NZFS around Banks Peninsula (Chapter Four), an area classified as at high risk of oil spills (Navigatus Consulting 2015). Prior to the study in Chapter Four, managers responding to a local spill would have been forced to rely on patchy and outdated NZFS distribution and abundance data (Boren et al. 2006b; Baird 2011; Allum and Maddigan 2012; Emami-Khoyi et al. 2016a), or anecdotal knowledge, when trying to mount a pinniped response. Pinnipeds are found near ports in other areas of New Zealand (Taylor et al. 1995; MacDiarmid and Lalas 2014), and in other countries (Bartholomew and Boolootian 1960; Stafford-Bell et al. 2012; Vincent et al. 2017), and regular monitoring is integral to informing local oil spill response planning. This is particularly true if the species in question is in a dynamic phase of population change and thus does not have a stable distribution and abundance, as has been shown for Banks Peninsula's NZFS population (Chapter 4), and/or when they are particularly hard to clean and rehabilitate (Ziccardi et al. 2015), which is true of NZFS in general.

When changes occur to wildlife population dynamics and human infrastructure simultaneously, the effects can be multifaceted. For example, Chapter Three demonstrated both that the overall number of live and/or dead NZFS ('NZFS incidents') on SH1 had increased since Boren et al. (2008), and that incident hotspot locations changed between 2012 – 2022 (Chapter 3). The former was likely due to the growth of the local NZFS population (Chapter 2) (Kazemi et al. 2016; Frangini et al. 2022), while the latter was likely driven by changes to SH1's infrastructure, most notably the construction of the sea wall at Ōhau Point. Another infrastructural change, the addition of motorcycle protection railing (MPR), appears to have prevented NZFS from accessing SH1 in places where it was installed post-earthquake. As such, NZTA installed another ca. 2 km of MPR at hotspots located in this thesis, in April 2024 (Appendix 3). This management measure should help reduce the number of NZFS incidents on SH1, with the associated benefit of lowering the risks to motorists (Spanowicz et al. 2020).

6.2.3 Gaps in our knowledge of NZFS ecology

Perhaps the most consistent thread running throughout this thesis is the overall lack of understanding of New Zealand's NZFS populations. This was demonstrated by the population monitoring gaps in both Kaikōura (Chapter 2) and Banks Peninsula (Chapter 4); the fact that the increased mortality of Kaikōura's NZFS on SH1 had gone unanalysed since Boren et al. (2008) (Chapter 3); and the limitations to any pinniped response plans on Banks Peninsula prior to the study in Chapter 4. However, the poor macro understanding of New Zealand's NZFS was most starkly illustrated in Chapter 5. The lack of consistent monitoring in most of the country, the variety of methodologies used over time (Taylor et al. 1995; Boren et al. 2006b; Baker et al. 2009; Hall et al. 2024 (Chapter 2)) and the scarcity of NZFS vital rate data from within New Zealand means the modelled abundance estimates provided in that chapter should be treated with caution.

There are several ecological implications to this lack of clarity. Firstly, NZFS are a top predator (Harcourt et al. 2002; Emami-Khoyi et al. 2016b), meaning that their relatively rapid recovery from exploitation (Dussex et al. 2016) will likely have already impacted New Zealand's marine ecosystems, as has been observed in other parts of the world (Rossi et al. 2021). Currently, throughout most of New Zealand, it would be impossible to confidently detect, let alone monitor or manage, NZFS-induced changes to marine ecosystems, due to the dearth of recent data on NZFS population parameters.

As well as impacting marine ecosystems, NZFS could be used to infer the health of these systems. This, however, would require a better understanding of the species' distribution and abundance. Due to their size, top predators such as NZFS are often the most noticeably impacted organisms when changes occur in the marine environment (Moore 2008; Bossart 2011), and this is particularly true for a species that gives birth and raises its offspring ashore, providing relatively easy opportunities for abundance monitoring (Berkson and DeMaster 1985). Studies of pinnipeds have been successfully used in the past to detect impacts on marine ecosystems ranging from climate induced change (Elorriaga-Verplancken et al. 2016, Hückstädt et al. 2017; Hendrix et al. 2021) to disease (Baily et al. 2016; Saab et al. 2023; Campagna et al. 2024). The latter is particularly relevant in the context of the spread of H5N1 avian influenza in pinnipeds (Ulloa et al. 2023; Campagna et al. 2024), and the disease's arrival in Antarctic and subantarctic regions from which it might reach New Zealand (Gartrell et al. 2024). Not knowing the locations of NZFS colonies, and/or being unaware of their typical abundance, may mean that managers have reduced ability to detect the presence of such pathogens early, and thus are on the back foot in trying to contain their spread. By updating the abundance estimates for Kaikōura and Banks Peninsula, new baselines have been established in this thesis for these regions which will permit future comparisons.

Finally, not monitoring NZFS population trajectories makes it harder for managers to identify colonies or populations in need of greater protection from threats. For example, the Ōhau Point New Zealand Fur Seal Sanctuary was implemented to protect this colony from tourist disturbance, but no such protections currently exist on the Kaikōura Peninsula. This is potentially because, until the 2022/23 survey (Chapter 2), only small numbers of pups (ca. 8) had been recorded in the literature (Boren et al. 2006b). However, it was estimated that 768 (± 15 SE) pups were produced on the Kaikōura Peninsula in 2022/23 (Chapter 2), with breeding substantially expanded from the one, spatially limited, site previously recorded (Boren et al. 2006b). The Kaikōura Peninsula is very popular with tourists (pers. obs.), and while NZFS colony expansion has clearly taken place over several years, it had not been scientifically documented. This means that the increased risk of deleterious interactions between humans and NZFS, including risks of harm to the animals (Acevedo-Gutiérrez et al. 2010) as well as zoonotic disease transfer (Donahoe et al. 2014), may not be adequately reflected in current management measures. The problem of data deficiency belying the need for additional management may exist in other parts of New Zealand but, by definition, cannot be confirmed. Similarly, in the marine environment, there are ongoing concerns about high levels of NZFS bycatch (MacKenzie et al. 2022). However, again, there are insufficient data to determine what impact this may be having on proximal colonies, or which colonies these individuals are most likely to be from. Given the suspected role of fisheries bycatch in the declines noted at some WCSI colonies (Roberts and Neale 2016), it is important that these interactions are understood so appropriate mitigation steps can be taken.

6.3 Management considerations and future research

Several management considerations and avenues for future research can be derived from the results presented in this thesis. These have been combined in this section because the most important management and conservation actions regarding NZFS require first improving our understanding of the species and its ecological relationships with New Zealand's marine and coastal ecosystems.

6.3.1 Collecting population parameter data

The most pressing management action regarding NZFS in New Zealand is improving our general understanding of the species' distribution and abundance. Lack of knowledge regarding pinniped populations' distributions, abundances and trajectories is common around the world (Kovacs et al. 2012). For NZFS, this data deficiency is at the heart of many of the management issues relevant to the species: assessing the adequacy of current protections, detecting where NZFS experience deleterious interactions with anthropogenic infrastructure and activities, and understanding how increased NZFS abundance may impact marine ecosystems. As such, the adoption of a coordinated approach to NZFS surveying is strongly suggested, as has occurred in Australia with Australian fur seals (*Arctocephalus pusillus doriferus*) (McIntosh et al. 2018a). Due to the substantial competition for conservation and research resources in New Zealand, this would likely take the form of regular assessments at baseline sites, as suggested in Chapter 5. To preserve the few longitudinal NZFS monitoring programs in New Zealand, these sites should include Kaikōura (Boren et al. 2006b; Gooday 2015; Chapter 2), the Otago/Catlins (MacDiarmid and Lalas 2014) and the three regularly monitored colonies on the WCSI (Taumaka, Cape Foulwind and Wekakura Point) (Roberts and Neale 2016). Cape Palliser is another site that should be monitored regularly, due to its status as the only substantial North Island NZFS colony. Regularising the timing and methodologies of NZFS surveys in New Zealand would enable managers to detect and react to changing population dynamics (Roberts and Neale 2016), and respond more efficiently to emerging threats, such as novel pathogens (Gartrell et al. 2024). Certain areas, such as New Zealand's subantarctic islands and parts of Fiordland, have been particularly neglected when it comes to NZFS population monitoring (Taylor 1982, 1992; Moore and Moffat 1990). While such locations are unlikely to be monitored regularly due to their relative inaccessibility, assessments in these areas should be prioritised. The findings from the two population studies in this thesis (Chapters 2 and 4) add to the body of evidence that mark-recapture studies provide higher pup estimates than direct counts at the same sites (Watson et al. 2009; Chilvers 2021) and thus the former are typically considered more accurate (Chilvers 2021). However, mark-recapture studies are also more expensive, time-consuming and disruptive to NZFS than direct counts (Watson et al. 2009). Direct counts can provide precise estimates of population abundance through time if executed consistently, meaning they can be used to assess trends (Watson et al. 2009). Additionally, the accuracy of direct counts can be improved by using calibration indices (Watson et al. 2009) such as those applied in Chapters 2 and 4, calculated through assessing the difference in the results derived from direct counts and mark-recapture at the same site. Given that topography appears to be the main factor differentiating the results of direct counts and mark recapture (Watson et al. 2009), such calibration indices should not require regular updating, unless there are significant changes to a colony habitat, such as in Kaikōura after the 2016 earthquake (Chapter 2). Thus, if the goal of these proposed future monitoring efforts was to monitor trends in NZFS abundance, surveyors could largely rely on consistently executed direct counts, after initial assessments using both methodologies have been conducted to calculate calibration indices. Improvements to remotely piloted aircraft (drones) technology offer other avenues to provide precise estimates of pinniped populations (Sorrell et al. 2019), while again reducing impacts on the study population and lessening the time and costs associated with both mark-recapture and direct counts.

Where NZFS colonies already exist and are expanding, habitat surveys may be beneficial to project where recolonisation may occur next. There is a good understanding of what constitutes suitable NZFS breeding colony habitat (Ryan et al. 1997; Bradshaw et al. 1999), and conducting habitat assessments would help both to direct census survey coverage and pre-empt potential management issues arising from continued NZFS recolonisation. Gaining a better understanding of immigration patterns in NZFS would be similarly beneficial, given the important role immigration has played in recolonisation thus far (Bradshaw et al. 2000b; Dussex et al. 2016).

Regular NZFS abundance and distribution monitoring at local and regional scales would also help managers begin to track the ecosystem impacts of continued NZFS recolonisation, by permitting ecosystem changes to be measured against changes to NZFS population dynamics.

There is also an important social dimension to this. There have long been calls for NZFS culls in New Zealand, mainly due to perceived impacts on fisheries (Lalas and Bradshaw 2001), which have mirrored similar complaints in South Australia, again derived from concerns raised by recreational and commercial anglers (Shaughnessy et al. 2018). Such a claim, made by Foveaux Strait fisheries, led to the last legal cull of NZFS in New Zealand in 1946, where 6,187 NZFS were killed due to concerns over reduced blue cod (*Parapercis colias*) harvests (Lalas and Bradshaw 2001). Ultimately, no blue cod remains were found in the stomach contents of the NZFS culled (Sorensen 1969). A cull of Cape fur seal (*Arctocephalus pusillus pusillus*) in South Africa was similarly unsuccessful, as reduced fur seal abundance led to a release of downwards pressure on predatory fish consuming the commercially desirable species, with negative impacts on the fisheries (Punt and Butterworth 1995). While only ca. 10% of NZFS' diet overlaps with species of commercial value, the biomass contribution of these species to NZFS diet is unknown (Emami-Khoyi et al. 2016b). As such, NZFS foraging ecology studies addressing this question, combined with a greater understanding of the species' distribution and abundance, would enable the debate around NZFS' impacts on recreational and commercial fisheries to be addressed empirically. Foraging ecology studies would need to take an ecosystem-based approach (Morishita 2008; Goldsworthy et al. 2003,13) to avoid any resulting management actions repeating the mistakes made in South Africa (Punt and Butterworth 1995). Additionally, any consideration of future controls on NZFS should also evaluate the impacts on New Zealand's eco-friendly reputation (Jackman et al. 2024) which plays an important role in marketing its lucrative tourism industry.

While this thesis has advanced knowledge of the terrestrial distribution of NZFS in Kaikōura and Banks Peninsula, we also need to understand the marine portions of NZFS distribution. Information on preferred foraging areas has been collected for populations of NZFS in

Australia (Page et al. 2005), and in parts of New Zealand (Harcourt et al. 2002). However, in many parts of New Zealand, there is either no empirical understanding, or only assumptions (Allum and Maddigan 2012), as to where NZFS forage. Without this part of the picture, managers cannot fully appreciate the impacts of marine based risks such as bycatch (Roberts and Neale 2016; Pavanato et al. 2023b), and predation, the latter of which may change as climate change impacts the distribution of species that predate upon NZFS (Bradford et al. 2020).

6.3.2 Protected areas for NZFS

Based on the discovery that the boundaries of the Ōhau Point New Zealand Fur Seal Sanctuary no longer encompass the bulk of the NZFS breeding distribution at this site (Chapter 2), potential changes to the existing sanctuary boundaries have been suggested to local managers and stakeholders during a legislative review process that began in 2024. Increasing the sanctuary's size to incorporate all areas where NZFS currently breed at Ōhau Point would enable the sanctuary to meet its primary objective of protecting NZFS from terrestrial anthropogenic disturbance as set out in its founding legislation, Kaikōura Te Tai o Marokura Marine Management Act 2014. The expansion of Kaikōura's NZFS population at other locations on the Greater Kaikōura Coastline, means that similar protective measures may be required elsewhere in the region. In particular, the Kaikōura Peninsula, which is very popular with tourists, is somewhere NZFS could experience frequent anthropogenic disturbance (Acevedo-Gutiérrez et al. 2010). Such measures would need to consider other interests, such as areas important to local Māori for traditional food gathering (mahinga kai), and existing eco-tourism enterprises. As eco-tourism continues to grow globally (Xu et al. 2023), managers in destinations popular with ecotourists, such as Kaikōura (Simmons and Fairweather 1998), will need to regularly review existing protection frameworks to minimise deleterious interactions between people and local wildlife (Acevedo-Gutiérrez et al. 2010). This is particularly true when wildlife populations are expanding in abundance and distribution, as the locations where humans and wildlife interact will likely change through time. There is a significantly lower need for terrestrial protections for NZFS on Banks Peninsula, as most colonies can only be accessed via private land, meaning they experience few to no terrestrial anthropogenic intrusions. The need for terrestrial NZFS sanctuaries in other parts of New Zealand is beyond the scope of this thesis, however, it would likely be impossible to judge whether such measures were required, in many cases, due to the lack of NZFS population data. Cape Palliser, on the south coast of the North Island, is one location where such measures may be required, due to the area's popularity with tourists and the ease of access for humans into the NZFS colony (pers. obs.). However, a mark-recapture study is first required here to gain a better understanding of the total population and its distribution.

The same issues arise when considering the need for marine space-based protections for NZFS. Currently, no marine sanctuaries targeted at NZFS exist in New Zealand, although some colonies may benefit from umbrella protections offered by marine reserves, such as the Tonga Island Marine Reserve in the Abel Tasman region (Bradshaw 1999). Space-based marine protections for pinnipeds have been shown to be effective in reducing bycatch in Australian sea lions (*Neophoca cinerea*) (Goldsworthy et al. 2022), and such measures exist in New Zealand, for example at the Auckland Islands, where all fishing is excluded within a 12 nautical mile radius of the coast due to concerns around New Zealand sea lion (*Phocarctos hookeri*) bycatch (Hamilton and Baker 2016). Importantly, New Zealand sea lions are listed as endangered by the IUCN (Chilvers 2015) and have experienced consistently declining pup production in their main breeding areas (Chilvers and Meyer 2017), neither of which is true of NZFS overall. However, marine sanctuary type protections, as well as other potential mitigation options (Quierolo et al. 2025) could be considered for NZFS populations if bycatch is deemed a population level threat, as might be true at WCSI colonies (Roberts and Neale 2016), however a full discussion of these recourses are beyond the scope of this thesis.

6.3.3 Interactions with human infrastructure

Several lessons for managing interactions between pinnipeds and anthropogenic infrastructure can be learned from the Kaikōura road study (Chapter 3). First, this study demonstrates the benefits of long-term data sets for monitoring the impacts of human activity and infrastructure on wildlife populations (Bejder et al. 2006; Brockie et al. 2009). This study had access to 10 years of data (2012 – 2022) collected by NZTA to analyse, and comparisons could be made with an earlier 10 years (1996 – 2005) analysed by Boren et al. (2008). As such, it was possible to determine not only that the number of NZFS incidents had increased over the long term, but also to identify spatial hotspots on SH1, and explore how their locations had changed through time. This final analysis then permitted comparisons between changing hotspot locations and road barrier types, leading to the assessment that MPR is likely to be effective in preventing NZFS accessing SH1, and the 2024 installation of more of this barrier type (Appendix 3). With these barriers now in place, it is important that a follow-up study is conducted to assess their efficacy (Ford et al. 2022). Chapter 3 also demonstrated how anthropogenic infrastructure can have complex consequences for wildlife. For example, the Ōhau Point seawall has had the positive impact of reducing NZFS road incidents north of Kaikōura. However, it has also reduced the width of the Ōhau Point colony, leading to breeding redistribution. If redistribution continues, potentially due to climate induced inundations at narrow sections of the colony (McLean et al. 2018), new NZFS hotspots may arise on SH1, if effective barriers are not in place. As well as continued data collection on NZFS incidents on SH1, we recommend that data collection regarding NZFS on the railway line running alongside the road is commenced, as it is known that NZFS are also killed on this piece of transport infrastructure (J. Weir, pers. comm.). NZFS have been killed on the road near Cape Palliser (pers. obs.), and it is likely that this problem will be replicated elsewhere in New Zealand in the future, particularly if the species continues to recolonise areas in the North Island, which has a considerably denser human population than the South Island (Dix 1993; Bouma et al. 2008). If the follow up research in Kaikōura determines that the additional MPR installation is successful, as preliminary observations indicate (J. Weir, pers. comm.), this approach could provide a blueprint for mitigating the problems of pinnipeds accessing roads elsewhere in New Zealand, and in other countries where such problems exist (Hatfield and Rathbun 1999).

Chapter 4 highlighted the need for pinniped plans to be incorporated into oil spill response planning. This is particularly true for fur seals, given how hard they are to clean and rehabilitate if oiled (Ziccardi et al. 2015), and for populations in high-risk areas, such as near ports (Battershill et al. 2016). Additionally, the continued expansion of commercial and tourist shipping into geographically remote regions, and areas with high ecological value (Cerveny et al. 2020; Lau et al. 2023), means that spill plans need to be devised for such locations as well, due to the increased spill risk associated with greater vessel traffic (Chilvers et al. 2021). As the optimum wildlife response during an oil spill involves preventing wildlife from becoming oiled (Wolfaardt et al. 2008; Nijkamp et al. 2014; Chilvers and Battley 2019; Hong et al. 2020), it is important that managers know the size and distribution of any NZFS populations in their region. This again highlights the need for regularised and consistent NZFS population monitoring. If NZFS continue to recolonise the North Island (Dix 1993; Bouma et al. 2008), the risks of populations being impacted by oil spills will rise substantially, as this is where most of New Zealand's larger ports are located. For countries like New Zealand, where there are no dedicated centres capable of rehabilitating large numbers of oiled pinnipeds, effective oil spill response options will likely focus on preventing oil reaching colonies where there are pups (Mearns et al. 1999), and hazing or deterring animals in the water away from slicks (Chilvers 2024). While a variety of hazing and deterrence techniques have been used on pinnipeds in non-oil spill scenarios, these have met with mixed success (Tidwell et al. 2021; Vetemaa et al. 2021; Lehtonen et al. 2022), and none have been trialled with NZFS. As such, research trials are needed to determine which are likely to be the most effective, and to evaluate potentially harmful impacts on both NZFS and other marine species (Simonis et al. 2020).

6.4 Concluding remarks

This thesis has updated our understanding of NZFS population dynamics in two ecologically important areas of New Zealand (Banks et al. 2002; De Leo et al. 2010), demonstrated threats NZFS face from anthropogenic infrastructure and activities, as well as natural disasters, and highlighted important gaps in our current nationwide knowledge of the species.

NZFS are continuing to expand in abundance and distribution around New Zealand, increasing the opportunities for interactions with humans, and impacting marine ecosystems already under stress due to macro forces such as climate change (Keegan et al. 2022). The contemporary lack of understanding around NZFS population dynamics in most parts of the country means that managers are currently not adequately positioned to identify and respond to new threats to the species, such as the H5N1 avian influenza pandemic, or to understand the wider impacts of the return of this top predator. As such, increasing our knowledge of important population parameters will help guide future management of a species with important ecological, cultural, and economic significance.

This study also highlights the need for longitudinal monitoring of wildlife populations to evaluate the impacts of both catastrophic events, such as natural disasters and oil spills, and evolving threats, such as the risks posed by proximity to anthropogenic infrastructure.

Improvements to remote-monitoring technologies, such as unmanned aerial vehicles, represent exciting new avenues for researchers seeking to monitor pinnipeds (Gooday et al. 2018; McIntosh et al. 2018b), and exploiting these opportunities will not only enable greater understanding of species such as NZFS, but also the changing marine ecosystems they inhabit.

Bibliography

Abra FD, Granziera BM, Huijser MP, et al (2019) Pay or prevent? Human safety, costs to society and legal perspectives on animal-vehicle collisions in São Paulo state, Brazil. PLoS One 14:e0215152. https://doi.org/10.1371/journal.pone.0215152

Acevedo P, Quirós-Fernández F, Casal J, Vicente J (2014) Spatial distribution of wild boar population abundance: Basic information for spatial epidemiology and wildlife management. Ecological Indicators 36:594–600. https://doi.org/10.1016/j.ecolind.2013.09.019

Acevedo-Gutiérrez A, Acedvedo L, Belonovich O, Boren L (2010) How effective are posted signs to regulate tourism? An example with New Zealand fur seals. Tourism in Marine Environments 7:39–41. https://doi.org/https://doi.org/10.3727/154427310X12826772784874

Acevedo-Gutiérrez A, Acevedo L, Boren L (2011) Efectos de la Presencia de Voluntarios en el Hostigamiento a Focas Finas de Nueva Zelanda. Conservation Biology 25:623–627. https://doi.org/10.1111/j.1523-1739.2010.01611.x

Aghababaei MT, Costello SB, Ranjitkar P (2020) Assessing operational performance of New Zealand's South Island road network after the 2016 Kaikoura Earthquake. International Journal Disaster Risk Reduction 47:101553. https://doi.org/10.1016/j.ijdrr.2020.101553

Agnew P (2024) Proceedings of the 13th Ōamaru Penguin Symposium 2023. New Zealand Journal of Zoology 51:436–451. https://doi.org/10.1080/03014223.2024.2332118

Alauddin M, Nghiemb HS (2010) Do Instructional Attributes pose Multicollinearity Problems? An Empirical Exploration. Economic Analysis and Policy 40:351–361. https://doi.org/10.1016/S0313-5926(10)50034-1

Alestra T, Gerrity S, Dunmore R, et al (2019) Rocky reef impacts of the Kaikōura earthquake. Quantification and monitoring of nearshore habitats and communities. Fisheries New Zealand, Tini a Tangaroa. Wellington, New Zealand.

Allen WJ, Helps FW, Molles LE, et al (2011) Factors affecting breeding success of the Flea Bay white-flippered penguin (*Eudyptula minor albosignata*) colony. New Zealand Journal of Ecology 35:199–208

Allum LL, Maddigan FW (2012) Unusual stability of diet of the New Zealand fur seal (*Arctocephalus forsteri*) at Banks Peninsula, New Zealand. New Zealand Journal of Marine and Freshwater Research 46:91–96. https://doi.org/10.1080/00288330.2011.604336

Almaraz P, Martínez F, Morales-Reyes Z, et al (2022) Long-term demographic dynamics of a keystone scavenger disrupted by human-induced shifts in food availability. Ecological Applications 32. https://doi.org/10.1002/eap.2579

Anderson AG (2002) The Media Politics of Oil Spills. Spill Science & Technology Bulletin 7:7–15. https://doi.org/10.1016/S1353-2561(02)00048-8

Anderson SS, Harwood J (1985) Time budgets and topography: how energy reserves and terrain determine the breeding behaviour of grey seals. Animal Behaviour 33:1343–1348. https://doi.org/10.1016/S0003-3472(85)80196-2

Anderson TK (2009) Kernel density estimation and K-means clustering to profile road accident hotspots. Accident Analysis & Prevention 41:359–364. https://doi.org/10.1016/j.aap.2008.12.014

Arnould J, Littnan C, Lento G (2000) First contemporary record of New Zealand fur seals *Arctocephalus forsteri* breeding in Bass Strait. Australian Mammalogy 22:57. https://doi.org/10.1071/AM00057

Atkinson S, Arnould JPY, Mashburn KL (2011) Plasma cortisol and thyroid hormone concentrations in pre-weaning Australian fur seal pups. General and Comparative Endocrinology 172:277–281. https://doi.org/10.1016/j.ygcen.2011.03.014

Augé AA, Chilvers BL, Mathieu R, Moore AB (2012) On-land habitat preferences of female New Zealand sea lions at Sandy Bay, Auckland Islands. Marine Mammal Science 28:620–637. https://doi.org/10.1111/j.1748-7692.2011.00515.x

Austin M (2007) Species distribution models and ecological theory: A critical assessment and some possible new approaches. Ecological Modelling 200:1–19. https://doi.org/10.1016/j.ecolmodel.2006.07.005

Baily JL, Foster G, Brown D, et al (2016) Salmonella infection in grey seals (*Halichoerus grypus*), a marine mammal sentinel species: pathogenicity and molecular typing of Salmonella strains compared with human and livestock isolates Environmental Microbiology 18:1078–1087. https://doi.org/10.1111/1462-2920.13219

Baird SJ (2011) New Zealand fur seals – summary of current knowledge New Zealand Aquatic Environment and Biodiversity Report No. 72. NIWA. Wellington, New Zealand.

Baker B, Jensz K, Cawthorn M, Cunningham R (2009) Census of New Zealand Fur Seals on the West Coast of New Zealand's South Island Report prepared for Deepwater Group Limited

Baker JD, Harting AL, Johanos TC, Littnan CL (2016) Estimating Hawaiian monk seal range-wide abundance and associated uncertainty. Endangered Species Research 31:317–324. https://doi.org/10.3354/esr00782

Baker PJ, Dowding C V., Molony SE, et al (2007) Activity patterns of urban red foxes (*Vulpes vulpes*) reduce the risk of traffic-induced mortality. Behavioural Ecology 18:716–724. https://doi.org/10.1093/beheco/arm035

Banks JC, Mitchell AD, Waas JR, Paterson AM (2002) An unexpected pattern of molecular divergence within the blue penguin (Eudyptula minor) complex

Barrett C, Hayes D (2019) Large-Scale Earthquake-Induced Landslide Repair Following New Zealand's Kaikoura Earthquakes. In: 70th Highway Geology Symposium

Bartholomew GA, Boolootian RA (1960) Numbers and Population Structure of the Pinnipeds on the California Channel Islands. Journal of Mammalogy 41: 366-375. https://doi.org/10.2307/1377495

Barton K, Booth K, Ward J, et al (1998) Visitor and New Zealand Fur Seal Interactions Along the Kaikoura Coast. Lincoln University.

Battershill CN, Ross PR, Schiel DR (2016) The MV Rena shipwreck: time-critical scientific response and environmental legacies. New Zealand Journal of Marine and Freshwater Research 50:173–182. https://doi.org/10.1080/00288330.2015.1134593

Baxter-Gilbert JH, Riley JL, Neufeld CJH, et al (2015) Road mortality potentially responsible for billions of pollinating insect deaths annually. Journal of Insect Conservation 19:1029–1035. https://doi.org/10.1007/s10841-015-9808-z

Baylis AMM, Page B, McKenzie J, Goldsworthy SD (2012) Individual foraging site fidelity in lactating New Zealand fur seals: Continental shelf vs. oceanic habitats. Marine Mammal Science 28:276–294. https://doi.org/10.1111/j.1748-7692.2011.00487.x

Baylis AMM, Page B, Peters K, et al (2005) The ontogeny of diving behaviour in New Zealand fur seal pups (*Arctocephalus forsteri*). Canadian Journal of Zoology 83:1149–1161. https://doi.org/10.1139/z05-097

Beaudry F, Demaynadier PG, Hunter ML (2010) Identifying Hot Moments in Road-Mortality Risk for Freshwater Turtles. Journal of Wildlife Management 74:152–159. https://doi.org/10.2193/2008-370

Beauplet G, Barbraud C, Chambellant M, Guinet C (2005) Interannual variation in the post-weaning and juvenile survival of subantarctic fur seals: Influence of pup sex, growth rate and oceanographic conditions. Journal of Animal Ecology 74:1160–1172. https://doi.org/10.1111/j.1365-2656.2005.01016.x

Beauplet G, Barbraud C, Dabin W, et al (2006) Age-specific survival and reproductive performances in fur seals: Evidence of senescence and individual quality. In: Oikos. pp 430–441

Bejder L, Samuels A, Whitehead H, et al (2006) Decline in Relative Abundance of Bottlenose Dolphins Exposed to Long-Term Disturbance. Conservation Biology 20:1791–1798. https://doi.org/10.1111/j.1523-1739.2006.00540.x

Bennison A, Byrne AMP, Lynton-Jenkins JG, et al (2024) Detection and spread of high pathogenicity avian influenza virus H5N1 in the Antarctic Region. bioRxiv, 2023-11.

Benoit-Bird KJ, Wursig B, Mfadden CJ (2004) Dusky dolphin (*Lagenorhynchus obscurus*) forgaing in two different habitats: active acoustic detection of dolphins and their prey. Marine Mammal Science 20:215–231. https://doi.org/10.1111/j.1748-7692.2004.tb01152.x

Benson JF, Mahoney PJ, Sikich JA, et al (2016) Interactions between demography, genetics, and landscape connectivity increase extinction probability for a small population of large carnivores in a major metropolitan area. Proceedings of the Royal Society B: Biological Sciences 283:20160957. https://doi.org/10.1098/rspb.2016.0957

Berkson JM, DeMaster DP (1985) Use of Pup Counts in Indexing Population Changes in Pinnipeds. Canadian Journal of Fisheries and Aquatic Sciences 42:873–879. https://doi.org/10.1139/f85-111

Bernstein AS, Ando AW, Loch-Temzelides T, et al (2022) The costs and benefits of primary prevention of zoonotic pandemics. Science Advances 8:4183. https://doi/10.1126/sciadv.abl4183

Berón-Vera B, Crespo EA, Raga JA, Pedraza SN (2004) *Uncinaria hamiltoni* (Nematoda: *Ancylostomatidae*) in South American Sea Lions, Otaria flavescens, From Northern Patagonia, Argentina. Journal of Parasitology 90:860–863. https://doi.org/10.1645/GE-182R

Bíl M, Andrášik R, Dul'a M, Sedoník J (2019) On reliable identification of factors influencing wildlife-vehicle collisions along roads. Journal of Environmental Management 237:297–304. https://doi.org/10.1016/j.jenvman.2019.02.076

Bíl M, Andrášik R, Janoška Z (2013) Identification of hazardous road locations of traffic accidents by means of kernel density estimation and cluster significance evaluation. Accident Analysis & Prevention 55:265–273. https://doi.org/10.1016/j.aap.2013.03.003

Bíl M, Andrášik R, Svoboda T, Sedoník J (2016) The KDE+ software: a tool for effective identification and ranking of animal-vehicle collision hotspots along networks. Landscape Ecology 31:231–237. https://doi.org/10.1007/s10980-015-0265-6

Bino G, Kingsford RT, Wintle BA (2020) A stitch in time – Synergistic impacts to platypus metapopulation extinction risk. Biological Conservation 242:108399. https://doi.org/10.1016/j.biocon.2019.108399

Blake DM, Stevenson J, Wotherspoon L, et al (2019) The role of data and information exchanges in transport system disaster recovery: A New Zealand case study. International Journal of Disaster Risk Reduction 39. https://doi.org/10.1016/j.ijdrr.2019.101124

Blanchet M-A, Vincent C, Womble JN, et al (2021) Harbour Seals: Population Structure, Status, and Threats in a Rapidly Changing Environment. Oceans 2:41–63. https://doi.org/10.3390/oceans2010003

Bland LM, Collen B, Orme CDL, Bielby J (2015) Predicting the conservation status of data-deficient species. Conservation Biology 29:250–259. https://doi.org/10.1111/cobi.12372

Bodkin JL, Ballachey BE, Dean TA, et al (2002) Sea otter population status and the process of recovery from the 1989 'Exxon Valdez' oil spill. Marine Ecology Progress Series241:237–253. https://doi.org/doi:10.3354/meps241237

Boots BN, Getis A (1988) Point Pattern Analysis. SAGE Publications Incorporated, Beverley Hills, CA.

Borda-de-Água L, Grilo C, Pereira HM (2014) Modeling the impact of road mortality on barn owl (*Tyto alba*) populations using age-structured models. Ecological Modelling 276:29–37. https://doi.org/10.1016/j.ecolmodel.2013.12.022

Boren LJ (2001) Assessing the Impact of Tourism on New Zealand (*Arctocephalus forsteri*). Master's thesis, University of Canterbury

Boren LJ (2005) New Zealand fur seals in the Kaikoura region: colony dynamics, maternal investment and health. Doctoral Dissertation, University of Canterbury

Boren LJ, Gemmell NJ, Barton KJ (2002) Tourist disturbance on New Zealand fur *seals Arctocephalus forsteri*. Australian Mammalogy 24:85–96. https://doi.org/https://doi.org/10.1071/AM02085

Boren LJ, Morrissey M, Gemmell NJ (2008) Motor vehicle collisions and the New Zealand fur seal in the Kaikoura region. Marine Mammal Science 24:235–238. https://doi.org/10.1111/j.1748-7692.2007.00172.x

Boren LJ, Morrissey M, Muller CG, Gemmell NJ (2006a) Entanglement of New Zealand fur seals in man-made debris at Kaikoura, New Zealand. Marine Pollution Bulletin 52:442–446. https://doi.org/10.1016/j.marpolbul.2005.12.003

Boren LJ, Muller CG, Gemmell NJ (2006b) Colony growth and pup condition of the New Zealand fur seal (*Arctocephalus forsteri*) on the Kaikoura coastline compared with other east coast colonies. Wildlife Research 33:497–505. https://doi.org/10.1071/WR05092

Bossart GD (2011) Marine mammals as sentinel species for oceans and human health. Veterinary Pathology 48:676–690. https://doi.org/10.1177/0300985810388525

Bouma S, Hickman G, Taucher D (2008) Abundance and reproduction of the New Zealand fur seal (*Arctocephalus forsteri*) along the West Coast of the Waikato Region, New Zealand. Journal of the Royal Society of New Zealand 38:89–96. https://doi.org/10.1080/03014220809510548

Boveng PL, Hiruki LM, Schwartz MK, Bengtson JL (1998) Population growth of antarctic fur seals: Limitation by a top predator, the leopard seal? Ecology 79:2863–2877. https://doi.org/10.1890/0012-9658(1998)079[2863:PGOAFS]2.0.CO;2

Bowen WD, Lidgard D (2013) Marine mammal culling programs: Review of effects on predator and prey populations. Mammal Review 43:207–220. https://doi.org/10.1111/j.1365-2907.2012.00217.x

Boyd C, Brooks TM, Butchart SHM, et al (2008) Spatial scale and the conservation of threatened species. Conservation Letters 1:37–43. https://doi.org/10.1111/j.1755-263X.2008.00002.x

Boyd IL, Croxall JP, Lunn NJ, Reid K (1995) Population Demography of Antarctic Fur Seals: The Costs of Reproduction and Implications for Life-Histories. Journal of Animal Ecology 64:505–518. https://doi.org/10.2307/5653

Boyd IL, Lunn NJ, Rothery P, D J P Croxall AN (1990) Age distribution of breeding female Antarctic fur seals in relation to changes in population growth rate. Canadian Journal of Zoology 68:2209–2213. https://doi.org/10.1139/z90-307

Bradford R, Patterson TA, Rogers PJ, et al (2020) Evidence of diverse movement strategies and habitat use by white sharks, *Carcharodon carcharias*, off southern Australia. Marine Biology 167. https://doi.org/10.1007/s00227-020-03712-y

Bradshaw CJA (1999) Population Dynamics and Colonisation Modelling of New Zealand Fur Seals. Doctoral Dissertation, University of Otago

Bradshaw CJA, Barker RJ, Harcourt RG, Davis LS (2003) Estimating survival and capture probability of fur seal pups using multistate mark-recapture models. Journal of Mammalogy 84:65–80. https://doi.org/10.1644/1545-1542(2003)084<0065:ESACPO>2.0.CO;2

Bradshaw CJA, Davis LS, Lalas C, Harcourt RG (2000a) Geographic and temporal variation in the condition of pups of the New Zealand fur seal (*Arctocephalus forsteri*): evidence for density dependence and differences in the marine environment. Journal of Zoology 252:41–51. https://doi.org/https://doi.org/10.1111/j.1469-7998.2000.tb00818.x

Bradshaw CJA, Lalas C, Thompson CM (2000b) Clustering of colonies in an expanding population of New Zealand fur seals (*Arctocephalus forsteri*). Journal of Zoology 250:105–112. https://doi.org/https://doi.org/10.1111/j.1469-7998.2000.tb00581.x

Bradshaw CJA, Thompson CM, Davis LS, Lalas C (1999) Pup density related to terrestrial habitat use by New Zealand fur seals. Canadian Journal of Zoology 77:1579–1586. https://doi.org/https://doi.org/10.1139/z99-125

Brock PM, Hall AJ, Goodman SJ, et al (2013) Immune Activity, Body Condition and Human-Associated Environmental Impacts in a Wild Marine Mammal. PLoS One 8:e67132. https://doi.org/10.1371/journal.pone.0067132

Brockie RE, Sadleir RMFS, Linklater WL (2009) Long-term wildlife road-kill counts in New Zealand. New Zealand Journal of Zoology36:123–134. https://doi.org/10.1080/03014220909510147

Brook BW, Bradshaw CJA (2006) Strength of evidence for density dependence in abundance time series of 1198 species. Ecology 87:1445–1451. https://doi.org/10.1890/0012658(2006)87[1445:SOEFDD]2.0.CO;2

Brough T, Rayment W, Dawson S (2019) Using a recreational grade echosounder to quantify the potential prey field of coastal predators. PLoS One 14. https://doi.org/10.1371/journal.pone.0217013

Burger J (1997) Effects of oiling on feeding behaviour of sanderlings and semipalmated plovers in New Jersey. Ornithological Applications 2:290–298. https://doi.org/https://doi.org/10.2307/1369935

Calvin K, Dasgupta D, Krinner G, et al (2023) IPCC, 2023: Climate Change 2023: Synthesis Report. Contribution of Working Groups I, II and III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, H. Lee and J. Romero (eds.)]. IPCC, Geneva, Switzerland.

Campagna C, Uhart M, Falabella V, et al (2024) Catastrophic mortality of southern elephant seals caused by H5N1 avian influenza. Marine Mammal Science 40:322–325. https://doi.org/10.1111/mms.13101

Campbell R, Holley D, Collins P, Armstrong S (2014) Changes in the abundance and distribution of the New Zealand fur seal (*Arctocephalus forsteri*) in Western Australia: Are they approaching carrying capacity? Australian Journal of Zoology 62:261–267. https://doi.org/10.1071/ZO14016

Campera M, Santini L, Balestri M, et al (2020) Elevation gradients of lemur abundance emphasise the importance of Madagascar's lowland rainforest for the conservation of endemic taxa. Mammal Review 50:25–37. https://doi.org/10.1111/mam.12172

Carey PW (1991) Resource-defense Polygyny and Male Territory Quality in the New Zealand Fur Seal. Ethology 88:63–79. https://doi.org/10.1111/j.1439-0310.1991.tb00263.x

Carey PW (1992) Fish prey species of the New Zealand fur seal (*Arctocephalus forsteri*, Lesson). New Zealand Journal of Ecology 16:41–46. https://www.jstor.org/stable/24053586

Carey PW (1998) New Zealand fur seals (*Arctocephalus forsteri*) at the Snares Islands: A stabilised population? New Zealand Journal of Marine and Freshwater Research 32:113–118. https://doi.org/10.1080/00288330.1998.9516810

Carome W, Slooten E, Rayment W, et al (2022) A long-term shift in the summer distribution of Hector's dolphins is correlated with an increase in cruise ship tourism. Aquatic Conservation: Marine and Freshwater Ecosystems 32:1660–1674. https://doi.org/10.1002/aqc.3881

Cassini MH, Fernández-Juricic E (2003) Costs and benefits of joining South American sea lion breeding groups: Testing the assumptions of a model of female breeding dispersion. Canadian Journal of Zoology 81:1154–1160. https://doi.org/10.1139/z03-098

Caswell H (2001) Matrix Population Models. Sinauer, New York

Caudron AK, Negro SS, Fowler M, et al (2009) Alternative mating tactics in the New Zealand fur seal (*Arctocephalus forsteri*): When non-territorial males are successful too. Australian Journal of Zoology 57:409–421. https://doi.org/10.1071/ZO09024

Cerveny LK, Miller A, Gende S (2020b) Sustainable cruise tourism in marine world heritage sites. Sustainability 12. https://doi.org/10.3390/su12020611

Chambellant M, Beauplet G, Guinet C, Georges JY (2003) Long-term evaluation of pup growth and preweaning survival rates in subantarctic fur seals, *Arctocephalus tropicalis*, on Amsterdam Island. Canadian Journal of Zoology 81:1222–1232. https://doi.org/10.1139/z03-114

Chapman DG (1952) Inverse, Multiple and Sequential Sample Censuses. Biometrics 8:286. https://doi.org/10.2307/3001864

Chen J, Zhang W, Wan Z, et al (2019) Oil spills from global tankers: Status review and future governance. Journal of Cleaner Production 227:20–32. https://doi.org/10.1016/j.jclepro.2019.04.020

Childerhouse S, Gales N (1998) Historical and modern distribution and abundance of the New Zealand sea lion *Phocarctos hookeri*. New Zealand Journal of Zoology 25:1–16. https://doi.org/10.1080/03014223.1998.9518131

Chilvers BL (2012) Population viability analysis of New Zealand sea lions, Auckland Islands, New Zealand's sub-Antarctics: Assessing relative impacts and uncertainty. Polar Biology 35:1607–1615. https://doi.org/10.1007/s00300-011-1143-6

Chilvers BL (2015) *Phocarctos hookeri*, New Zealand Sea Lion THE IUCN RED LIST OF THREATENED SPECIES. https://doi.org/10.2305/IUCN.UK.2015-2.RLTS.T17026A1306343.en

Chilvers BL (2017) New Zealand Marine Oiled Wildlife Risk Assessment 2017. Palmerston North, New Zealand.

Chilvers BL (2018) New Zealand Fur Seal*: Arctocephalus forsteri*. In: Encyclopedia of Marine Mammals. Academic Press., pp 632–634

Chilvers BL (2019) Whisker stable isotope values indicate long-term foraging strategies for female New Zealand sea lions. Endangered Species Research 38:55–66. https://doi.org/10.3354/ESR00934

Chilvers BL (2021) Pup numbers, estimated population size, and monitoring of New Zealand fur seals in Doubtful/Pateā, Dusky and Breaksea Sounds, and Chalky Inlet, Fiordland, New Zealand 2021. New Zealand Journal of Marine and Freshwater Research 57:75–87. https://doi.org/10.1080/00288330.2021.1962921

Chilvers BL (2024) Techniques for hazing and deterring birds during an oil spill. Marine Pollution Bulletin 201:116276. https://doi.org/10.1016/j.marpolbul.2024.116276

Chilvers BL, Battley PF (2019) Species prioritization index for oiled wildlife response planning in New Zealand. Marine Pollution Bulletin 149. https://doi.org/10.1016/j.marpolbul.2019.110529

Chilvers BL, Dobbins ML (2021) Behavioural plasticity and population connectivity: Contributors to the establishment of new pinniped breeding colonies. Aquatic Conservation: Marine and Freshwater Ecosystems 31:2217–2228. https://doi.org/10.1002/aqc.3603

Chilvers BL, Goldsworthy SD (2015) *Arctocephalus forsteri*. The IUCN Red List of Threatened Species 2015: e.T41664A45230026. https://dx.doi.org/10.2305/IUCN.UK.2015-2.RLTS.T41664A45230026.en

Chilvers BL, Hiscock JA (2019) Significant decline of endangered Antipodes Island penguins: Is extreme weather an additional impact? Aquatic Conservation: Marine and Freshwater Ecosystems 29:546–553. https://doi.org/10.1002/aqc.3034

Chilvers BL, McClelland PJ (2023) Lessons Learned for Pre-Emptive Capture Management as a Tool for Wildlife Conservation during Oil Spills and Eradication Events. Animals 13. https://doi.org/10.3390/ani13050833

Chilvers BL, Meyer S (2017) Conservation needs for the endangered New Zealand sea lion, *Phocarctos hookeri*. Aquatic Conservation: Marine and Freshwater Ecosystems 27:846–855. https://doi.org/10.1002/aqc.2742

Chilvers BL, Morgan KJ, White BJ (2021) Sources and reporting of oil spills and impacts on wildlife 1970–2018. Environmental Science and Pollution Research 28:754–762. https://doi.org/10.1007/s11356-020-10538-0

Chilvers BL, Ruoppolo V (2023) Planning for an offshore oiled wildlife response: case studies from New Zealand and Brazil. Environmental Science and Pollution Research 30:54351–54361. https://doi.org/10.1007/s11356-023-26440-4

Chilvers BL, Wilkinson IS (2008) Philopatry and site fidelity of New Zealand sea lions (*Phocarctos hookeri*). Wildlife Research 35:463. https://doi.org/10.1071/WR07053

Chilvers BL, Wilkinson IS, Childerhouse S (2007) New Zealand sea lion, *Phocarctos hookeri*, pup production—1995 to 2006. New Zealand Journal of Marine and Freshwater Research 41:205–213. https://doi.org/10.1080/00288330709509909

Christie KS, Gilbert SL, Brown CL, et al (2016) Unmanned aircraft systems in wildlife research: Current and future applications of a transformative technology. Frontiers in Ecology and the Environment 14:241–251. https://doi.org/10.1002/fee.1281

Ciocheti G, de Assis JC, Ribeiro JW, Ribeiro MC (2017) Highway widening and underpass effects on vertebrate road mortality. Biotropica 49:765–769. https://doi.org/10.1111/btp.12480

Cipro CVZ, Bustamante P, Taniguchi S, Montone RC (2012) Persistent organic pollutants and stable isotopes in pinnipeds from King George Island, Antarctica. Marine Pollution Bulletin 64:2650–2655. https://doi.org/10.1016/j.marpolbul.2012.10.012

Clumpner C, Callahan B (2014) Optimizing the value of near misses in wildlife response preparedness: The Kulluk Incident. In: International Oil Spill Conference Proceedings. American Petroleum Institute., pp 2288–2294

Clutton-Brock T, Sheldon BC (2010) Individuals and populations: the role of long-term, individual-based studies of animals in ecology and evolutionary biology. Trends in Ecology & Evolution 25:562–573. https://doi.org/10.1016/j.tree.2010.08.002

Condit R, Reiter J, Morris PA, et al (2014) Lifetime survival rates and senescence in northern elephant seals. Marine Mammal Science 30:122–138. https://doi.org/10.1111/mms.12025

Conway R, Duncan C, Foster RA, et al (2022) Histologic lesions in placentas of Northern fur seals (*Callorhinus ursinus*) from a population with high placental prevalence of Coxiella burnetii. Journal of Wildlife Diseases 58:333–340. https://doi.org/10.7589/JWD-D-21-00037

Cornick LA, Neill W, Grant WE (2006) Assessing competition between Steller sea lions and the commercial groundfishery in Alaska: A bioenergetics modelling approach. Handbook of Environmental Chemistry, Volume 5: Water Pollution 199:107–114. https://doi.org/10.1016/j.ecolmodel.2006.07.003

Costa DP, Trillmich F (1988) Mass Changes and Metabolism during the Perinatal Fast: A Comparison between Antarctic (*Arctocephalus gazella*) and Galápagos Fur Seals (Arctocephalus galapagoensis). Physiological Zoology 61:160–169. https://doi.org/10.1086/physzool.61.2.30156147

Cowling M, Kirkwood R, Boren LJ, et al (2015) The effects of vessel approaches on the New Zealand fur seal (*Arctocephalus forsteri*) in the bay of plenty, New Zealand. Marine Mammal Science 31:501–519. https://doi.org/10.1111/mms.12171

Cowling M, Kirkwood R, Boren LJ, Scarpaci C (2014) The effects of seal-swim activities on the New Zealand fur seal (*Arctophoca australis forsteri*) in the Bay of Plenty, New Zealand, and recommendations for a sustainable tourism industry. Marine Policy 45:39–44. https://doi.org/10.1016/j.marpol.2013.11.010

Crain BJ, Tremblay RL, Ferguson JM (2019) Sheltered from the storm? Population viability analysis of a rare endemic under periodic catastrophe regimes. Population Ecology 61:74–92. https://doi.org/10.1002/1438-390X.1002

Crawley MC, Wilson GJ (1976) The natural history and behaviour of the New Zealand fur seal (*Arctocephalus forsteri*). Biological society, Victoria University of Wellington.

Curtin S (2003) Whale-Watching in Kaikoura: Sustainable Destination Development? Journal of Ecotourism 2:173–195. https://doi.org/10.1080/14724040308668143

Cuthbert RJ (2019) A reassessment of population size and trends of Hutton's shearwater following the 2016 Kaikōura earthquake and outlook for the species' management New Zealand Aquatic Environment and Biodiversity Report No. 209

D'Amico M, Román J, de los Reyes L, Revilla E (2015) Vertebrate road-kill patterns in Mediterranean habitats: Who, when and where. Biological Conservation 191:234–242. https://doi.org/10.1016/j.biocon.2015.06.010

Davis KJ (2022) Managed culls mean extinction for a marine mammal population when combined with extreme climate impacts. Ecological Modelling 473:110122. https://doi.org/10.1016/j.ecolmodel.2022.110122

Davison R, Stadman M, Jongejans E (2019) Stochastic effects contribute to population fitness differences. Ecological Modelling 408:108760. https://doi.org/10.1016/j.ecolmodel.2019.108760

De Leo FC, Smith CR, Rowden AA, et al (2010) Submarine canyons: hotspots of benthic biomass and productivity in the deep sea. Proceedings of the Royal Society B: Biological Sciences 277:2783–2792. https://doi.org/10.1098/rspb.2010.0462

De Villiers DJ, Roux JP (1992) Mortality of newborn pups of the South African fur seal *Arctocephalus pusillus pusillus* in Namibia. South African Journal of Marine Science 12:881–889. https://doi.org/10.2989/02577619209504749

Dean WRJ, Seymour CL, Joseph GS, Foord SH (2019) A Review of the Impacts of Roads on Wildlife in Semi-Arid Regions. Diversity (Basel) 11:81. https://doi.org/10.3390/d11050081

Delaney KS, Riley SPD, Fisher RN (2010) A rapid, strong, and convergent genetic response to urban habitat fragmentation in four divergent and widespread vertebrates. PLoS One 5:1–11. https://doi.org/10.1371/journal.pone.0012767

Denneboom D, Bar-Massada A, Shwartz A (2021) Factors affecting usage of crossing structures by wildlife – A systematic review and meta-analysis. Science of the Total Environment 777:146061. https://doi.org/10.1016/j.scitotenv.2021.146061

Department of Conservation (2022) Police have "solid lead" in Kaikōura seal killing case - DOC. Retrieved from: https://www.stuff.co.nz/environment/300658609/police-have-solid-lead-in-kaikura-seal-killing-case--doc

Department of Conservation (2023) Pup production estimates. Retrieved from: https://furseals.dragonfly.co.nz/pup-production

Descamps S, Boutin S, Berteaux D, et al (2008) Age-specific variation in survival, reproductive success and offspring quality in red squirrels: evidence of senescence. Oikos 117:1406–1416. https://doi.org/10.1111/j.2008.0030-1299.16545.x

Dickie GS, Dawson SM (2003) Age, growth, and reproduction in New Zealand fur seals. Marine Mammal Science 19:173–185. https://doi.org/10.1111/j.1748-7692.2003.tb01100.x

Digby A, Eason D, Catalina A, et al (2023) Hidden impacts of conservation management on fertility of the critically endangered kākāpō. PeerJ 11:e14675. https://doi.org/10.7717/peerj.14675

Dimitrakiev D, Dachev Y, Milev D (2020) Impact Of The Dispersants On The Marine Environment. International Journal of Scientific & Technology Research 9:710–712

Dix B (1993) A new record this century of a breeding colony in the North Island for the New Zealand fur seal Arctocephalus forsteri. Journal of the Royal Society of New Zealand 23:1–4. https://doi.org/10.1080/03036758.1993.10721212

do Rosário IT, Mathias M da L (2007) Post-fire recolonisation of a montado area by the endangered Cabrera vole (*Microtus cabrerae*). International Journal of Wildland Fire 16:450. https://doi.org/10.1071/WF06096

Donahoe SL, Rose K, Šlapeta J (2014) Multisystemic toxoplasmosis associated with a type II-like Toxoplasma gondii strain in a New Zealand fur seal (*Arctocephalus forsteri*) from New South Wales, Australia. International Journal of Wildland Fire 205:347–353. https://doi.org/10.1016/j.vetpar.2014.07.022

Drews C (1995) Road kills of animals by public traffic in Mikumi National Park, Tanzania, with notes on baboon mortality. African Journal of Ecology 33:89–100. https://doi.org/10.1111/j.1365-2028.1995.tb00785.x

Dudley IF, Coates PS, Prochazka BG, et al (2022) Maladaptive nest-site selection and reduced nest survival in female sage-grouse following wildfire. Ecosphere 13. https://doi.org/10.1002/ecs2.4282

Duignan P (2000) Diseases of cetaceans and pinnipeds. In: Marine Wildlife: the Fabian Fay Course for Veterinarians. Post Graduate Foundation in Veterinary Science, Proceedings 335. University of Sydney, Gold Coast, Australia, pp 419–447

Dunstan E, Funnell O, McLelland J, et al (2021) An Analysis of Demographic and Triage Assessment Findings in Bushfire-Affected Koalas (*Phascolarctos cinereus*) on Kangaroo Island, South Australia, 2019–2020. Animals 11:3237. https://doi.org/10.3390/ani11113237

Duputel Z, Rivera L (2017) Long-period analysis of the 2016 Kaikoura earthquake. Physics of the Earth and Planetary Interiors 265:62–66. https://doi.org/10.1016/j.pepi.2017.02.004

Dussex N, Robertson BC, Salis AT, et al (2016) Low spatial genetic differentiation associated with rapid recolonization in the New Zealand fur seal *Arctocephalus forsteri*. Journal of Heredity 107:581–592. https://doi.org/10.1093/jhered/esw056

Eacker DR, Lukacs PM, Proffitt KM, Hebblewhite M (2017) Assessing the importance of demographic parameters for population dynamics using Bayesian integrated population modeling. Ecological Applications 27:1280–1293. https://doi.org/10.1002/eap.1521

Egan AL, Rolfe J, Cassells S, Chilvers BL (2023) Recreational impacts of an oil spill in a world heritage area: a combined travel cost and contingent behaviour method approach. Journal of Environmental Economics and Policy 1–17. https://doi.org/10.1080/21606544.2023.2248951

Elorriaga-Verplancken FR, Sierra-Rodríguez GE, Rosales-Nanduca H, et al (2016) Impact of the 2015 El Niño-southern oscillation on the abundance and foraging habits of Guadalupe fur seals and California sea lions from the San Benito Archipelago, Mexico. PLoS One 11:e0155034. https://doi.org/10.1371/journal.pone.0155034

Emami-Khoyi A, Hartley DA, Paterson AM, et al (2016a) Mitochondrial DNA structure and colony expansion dynamics of New Zealand fur seals (*Arctocephalus forsteri*) around Banks Peninsula. New Zealand Journal of Zoology 43:322–335. https://doi.org/10.1080/03014223.2016.1179649

Emami-Khoyi A, Hartley DA, Paterson AM, et al (2016b) Identifying prey items from New Zealand fur seal (*Arctocephalus forsteri*) faeces using massive parallel sequencing. Conservation Genetics Resources 8:343–352. https://doi.org/10.1007/s12686-016-0560-9

Emami-Khoyi A, Paterson AM, Hartley DA, et al (2018) Mitogenomics data reveal effective population size, historical bottlenecks, and the effects of hunting on New Zealand fur seals (*Arctocephalus forsteri*). Mitochondrial DNA Part a 29:567–580. https://doi.org/10.1080/24701394.2017.1325478

Environment Canterbury (2024) Coastal pollution response. In: https://www.ecan.govt.nz/your-region/your-environment/coastal-environment/coastal-pollution-response/ (Date accessed: 21/11/2024).

Epps CW, Palsbøll PJ, Wehausen JD, et al (2005) Highways block gene flow and cause a rapid decline in genetic diversity of desert bighorn sheep. Ecology Letters 8:1029–1038. https://doi.org/10.1111/j.1461-0248.2005.00804.x

Erdsack N, Dehnhardt G, Hanke W (2013) Coping with Heat: Function of The Natal Coat of Cape Fur Seal (*Arctocephalus Pusillus Pusillus*) Pups in Maintaining Core Body Temperature. PLoS One 8:e72081. https://doi.org/10.1371/journal.pone.0072081

Erritzoe J, Mazgajski TD, Rejt Ł (2003) Bird Casualties on European Roads — A Review. Acta Ornithologica 38:77–93. https://doi.org/10.3161/068.038.0204

Fan J, Zhang F, Zhao D, Wang J (2015) Oil Spill Monitoring Based on SAR Remote Sensing Imagery. Aquatic Procedia 3:112–118. https://doi.org/10.1016/j.aqpro.2015.02.234

Fan Y, Rui X, Zhang G, et al (2021) Feature merged network for oil spill detection using sar images. Remote Sens (Basel) 13:3174. https://doi.org/10.3390/rs13163174

Ferguson SH, Young BG, Yurkowski DJ, et al (2017) Demographic, ecological, and physiological responses of ringed seals to an abrupt decline in sea ice availability. PeerJ 5:e2957. https://doi.org/10.7717/peerj.2957

Filkov AI, Ngo T, Matthews S, et al (2020) Impact of Australia's catastrophic 2019/20 bushfire season on communities and environment. Retrospective analysis and current trends. Journal of Safety Science and Resilience 1:44–56. https://doi.org/10.1016/j.jnlssr.2020.06.009

Fleming J, Moore JF, Waddle H, et al (2022) Range-Wide Population Projections for Northern Red-Bellied Cooters (*Pseudemys rubriventris*). Journal of Herpetology 56. https://doi.org/10.1670/21-065

Foo D, McMahon C, Hindell M, et al (2019) Influence of shelf oceanographic variability on alternate foraging strategies in long-nosed fur seals. Marine Ecology Progress Series 615:189–204. https://doi.org/10.3354/meps12922

Forcada J, Hoffman JI, Gimenez O, et al (2023) Ninety years of change, from commercial extinction to recovery, range expansion and decline for Antarctic fur seals at South Georgia. Global Change Biology 29:6867–6887. https://doi.org/10.1111/gcb.16947

Ford AT, Dorsey B, Lee TS, Clevenger AP (2022) A before-after-control-impact study of wildlife fencing along a highway in the Canadian Rocky Mountains. Frontiers in Conservation Science 3. https://doi.org/10.3389/fcosc.2022.935420

Frangini L, Sterrer U, Franchini M, et al (2022) Stay home, stay safe? High habitat suitability and environmental connectivity increases road mortality in a colonizing mesocarnivore. Landscape Ecology 37:2343–2361. https://doi.org/10.1007/s10980-022-01491-z

Fraser GS, Robertson GJ, Stenhouse IJ, Ellis JI (2022) Estimating the numbers of aquatic birds affected by oil spills: pre-planning, response, and post-incident considerations. Environmental Reviews 30:323–341. https://doi.org/10.1139/er-2021-0121

Frisman EYa, Skaletskaya EI, Kuzin AE (1982) A mathematical model of the population dynamics of a local Northern fur-seal herd. Ecological Modelling 16:151–172. https://doi.org/10.1016/0304-3800(82)90006-0

Fritt-Rasmussen J, Wegeberg S, Lassen P, et al (2023) Coastline in-situ burning of oil spills, analysis of a Greenland field experiment. The Journal of Hazardous Materials 441:129976. https://doi.org/10.1016/j.jhazmat.2022.129976

Frodella W, Rossi G, Tanteri L, et al (2023) A Decade from the Costa Concordia Shipwreck: Lesson Learned on the Contribution of Infrared Thermography during the Maritime Salvage Operations. Remote Sensing (Basel) 15:1347. https://doi.org/10.3390/rs15051347

Fujiwara M, Diaz-Lopez J (2017) Constructing stage-structured matrix population models from life tables: Comparison of methods. PeerJ 2017:e3971. https://doi.org/10.7717/peerj.3971

Fulham M, Power M, Gray R (2018) Comparative ecology of Escherichia coli in endangered Australian sea lion (*Neophoca cinerea*) pups. Infection, Genetics and Evolution 62:262–269. https://doi.org/10.1016/j.meegid.2018.05.002

Funayama K, Hines E, Davis J, Allen S (2013) Effects of sea-level rise on northern elephant seal breeding habitat at Point Reyes Peninsula, California. Aquatic Conservation: Marine and Freshwater Ecosystems 23:233–245. https://doi.org/10.1002/aqc.2318

Furlong KP, Herman M (2017) Reconciling the deformational dichotomy of the 2016 *M w* 7.8 Kaikoura New Zealand earthquake. Geophysical Research Letters 44:6788–6791. https://doi.org/10.1002/2017GL074365

Gales NJ (1991) New Zealand Fur Seals and Oil: An Overview of Assessment, Treatment, Toxic Effects and Survivorship. The 1991 Sanko Harvest Oil Spill. Mount Pleasant

Gales NJ, Fletcher DJ (1999) Abundance, distribution and status of the New Zealand sea lion, *Phocarctos hookeri*. Wildlife Research 26:35–52. https://doi.org/10.1071/WR98022

Gales NJ, Haberley B, Collins P (2000) Changes in the abundance of New Zealand fur seals, *Arctocephalus forsteri*, in Western Australia. Wildlife Research 27:165. https://doi.org/10.1071/WR99027

Galimberti F, Sanvito S, Boitani L, Fabiani A (2001) Viability of the southern elephant seal population of the Falkland Islands. Animal Conservation 4:81–88. https://doi.org/10.1017/S1367943001001093

Gallo-Reynoso JP, Égido-Villarreal J, Martínez-Villalba GL (2011) Reaction of fin whales *balaenoptera physalus* to an earthquake. Bioacoustics 20:317–329. https://doi.org/10.1080/09524622.2011.9753654

Garlepp L, Logan M, Kirkwood R (2014) Behavioral responses of Australian fur seals (*Arctocephalus pusillus doriferus*) to environmental variations. Marine Mammal Science 30:978–993. https://doi.org/10.1111/mms.12094

Garrah E, Danby RK, Eberhardt E, et al (2015) Hot Spots and Hot Times: Wildlife Road Mortality in a Regional Conservation Corridor. Journal of Environmental Management 56:874–889. https://doi.org/10.1007/s00267-015-0566-1

Garrott RA, Eberhardt LL, Burn DM (1993) Mortality of sea otters in Prince William Sound following the Exxon Valdez oil spill. Marine Mammal Science 9:343–359. https://doi.org/10.1111/j.1748-7692.1993.tb00468.x

Gartrell BD, Jolly MJ, Hunter SA (2024) The risks and consequences of a high pathogenicity avian influenza outbreak in Aotearoa New Zealand. New Zealand Veterinary Journal 72:63–65. https://doi.org/10.1080/00480169.2023.2294915

Gazo M, Aparicio F, Cedenilla MA, et al (2000) Pup survival in the Mediterranean monk seal (*Monachus monachus*) colony at Cabo Blanco Peninsula (Western Sahara-Mauritania). Marine Mammal Science 16:158–168. https://doi.org/10.1111/j.1748-7692.2000.tb00910.x

Gerber LR, Hilborn R (2001) Catastrophic events and recovery from low densities in populations of otariids: Implications for disk of extinction. Mammal Review 31:131–150. https://doi.org/10.1046/j.1365-2907.2001.00081.x

Ghadirian O, Moradi H, Madadi H, et al (2019) Identifying noise disturbance by roads on wildlife: a case study in central Iran. SN Applied Sciences 1:1–11. https://doi.org/10.1007/s42452-019-0838-0

Ghaly AE, Dave D (2011) Remediation Technologies for Marine Oil Spills: A Critical Review and Comparative Analysis. American Journal of Environmental Sciences 7:423–440. https://doi.org/10.3844/ajessp.2011.424.440

Gibbens J, Arnould JPY (2009) Age-specific growth, survival, and population dynamics of female Australian fur seals. Canadian Journal of Zoology 87:902–911. https://doi.org/10.1139/Z09-080

Gibbs NJ, Dunlop RA, Gibbs EJ, et al (2018) The potential beginning of a postwhaling recovery in New Zealand humpback whales (*Megaptera novaeangliae*). Marine Mammal Science 34:499–513. https://doi.org/10.1111/mms.12468

Girard F, Fisher CR (2018) Long-term impact of the Deepwater Horizon oil spill on deep-sea corals detected after seven years of monitoring. Biological Conservation 225:117–127. https://doi.org/10.1016/j.biocon.2018.06.028

Goldsworthy SD (1994) Breeding Biology and Haul-out Pattern of the New Zealand Fur Seal, *Arctocephalus forsteri*, at Cape Gantheaume, South Australia. Wildlife Research 21:365–76. https://doi.org/10.1071/WR9940365

Goldsworthy SD, Bulman C, He X et al (2003). Trophic interactions between marine mammals and Australian fisheries: an ecosystem approach. In Marine mammals: Fisheries, tourism and management issues (Vol. 2006, pp. 62-99). CSIRO Publishing Melbourne.

Goldsworthy SD, Gales N (2008) The IUCN Red List of Threatened Species. Version 2014.2. IUCN SSC Pinniped Specialist Group. Retrieved from: www.iucnredlist.org.

Goldsworthy SD, Page B (2007) A risk-assessment approach to evaluating the significance of seal bycatch in two Australian fisheries. Biological Conservation 139:269–285. https://doi.org/10.1016/j.biocon.2007.07.010

Goldsworthy SD, Page B, Hamer DJ, et al (2022) Assessment of Australian Sea Lion Bycatch Mortality in a Gillnet Fishery, and Implementation and Evaluation of an Effective Mitigation Strategy. Frontiers in Marine Science 9. https://doi.org/10.3389/fmars.2022.799102

Goldsworthy SD, Page B, Rogers PJ, et al (2013) Trophodynamics of the eastern Great Australian Bight ecosystem: Ecological change associated with the growth of Australia's largest fishery. Ecological Modelling 255:38–57. https://doi.org/10.1016/j.ecolmodel.2013.01.006

Goldsworthy SD, Shaughnessy PD (1994) Breeding biology and haul-out pattern of the New Zealand fur seal, *Arctocephalus forsteri*, at Cape Gantheaume, South Australia. Wildlife Research 21:365. https://doi.org/10.1071/WR9940365

Gonçalves LO, Alvares DJ, Teixeira FZ, et al (2018) Reptile road-kills in Southern Brazil: Composition, hot moments and hotspots. Science of the Total Environment 615:1438–1445. https://doi.org/10.1016/j.scitotenv.2017.09.053

González-Gallina A, Hidalgo-Mihart MG, Castelazo-Calva V (2018) Conservation implications for jaguars and other neotropical mammals using highway underpasses. PLoS One 13:ee0206614. https://doi.org/10.1371/journal.pone.0206614

Gooday OJ (2016) New Zealand fur seal (*Arctocephalus forsteri*) pup behaviour and an assessment of novel non-invasive population survey methods. Masters Dissertation, University of Canterbury.

Gooday OJ, Goldstien S (2018) Spatial assessment and status of New Zealand fur seal (*Arctocephalus forsteri*) colonies along the Kaikōura coastline in relation to the November 2016 Kaikōura earthquakes.

Gooday OJ, Key N, Goldstien S, Zawar-Reza P (2018) An assessment of thermal-image acquisition with an unmanned aerial vehicle (Uav) for direct counts of coastal marine mammals ashore. Journal of Unmanned Vehicle System 6:100–108. https://doi.org/10.1139/juvs-2016-0029

Govindarajulu P, Altwegg R, Anholt BR (2005) Matrix model investigation of invasive species control: Bullfrogs on Vancouver island. Ecological Applications 15:2161–2170. https://doi.org/10.1890/05-0486

Granquist SM, Sigurjonsdottir H (2014) The effect of land based seal watching tourism on the haul-out behaviour of harbour seals (*Phoca vitulina*) in Iceland. Applied Animal Behaviour Science 156:85–93. https://doi.org/10.1016/j.applanim.2014.04.004

Green R, Lambert C, Watts C, et al (2018) Development and Testing of a Modular Rockfall Protection Wall to Mitigate Earthquake-Induced Slope Hazards. In: 69th Highway Geology Symposium

Grilo C, Koroleva E, Andrášik R, et al (2020) Roadkill risk and population vulnerability in European birds and mammals. Frontiers in Ecology and the Environment 18:323–328. https://doi.org/10.1002/fee.2216

Grubesic TH, Nelson JR, Wei R (2019) A strategic planning approach for protecting environmentally sensitive coastlines from oil spills: Allocating response resources on a limited budget. Marine Policy 108: https://doi.org/10.1016/j.marpol.2019.103549

Guerra M, Dawson S, Sabadel A, et al (2020) Changes in habitat use by a deep-diving predator in response to a coastal earthquake. Deep Sea Research Part I: Oceanographic Research Papers Pap 158:103226. https://doi.org/10.1016/j.dsr.2020.103226

Guerra M, Hickmott L, van der Hoop J, et al (2017) Diverse foraging strategies by a marine top predator: Sperm whales exploit pelagic and demersal habitats in the Kaikōura submarine canyon. Deep Sea Research Part I: Oceanographic Research Papers Pap 128:98–108. https://doi.org/10.1016/j.dsr.2017.08.012

Guo P, Han Z jun, Ran H liu, et al (2019) Co-seismic surface rupture of Papatea fault and reactivation mechanism of the Clarence landslide during the 2016 M w7.8 Kaikoura earthquake, New Zealand. Bulletin of Engineering Geology and the Environment 78:3055–3068. https://doi.org/10.1007/s10064-018-1338-y

Guthery FS, Shaw JH (2013) Density dependence: Applications in wildlife management. Journal of Wildlife Management 77:33–38. https://doi.org/10.1002/jwmg.450

Haase T (2004) The determinants of weaning in the New Zealand fur seal. Doctoral Dissertation, La Trobe University, Victoria

Hall AA, Chilvers BL, Weir JS et al (2023) Post-earthquake highway reconstruction: Impacts and mitigation opportunities for New Zealand pinniped population. Ocean & Coastal Management 245:106851. https://doi.org/10.1016/j.ocecoaman.2023.106851

Hall AA, Chilvers BL, Weir JS, Boren LJ (2024) Earthquake impacts on a protected pinniped in New Zealand. Aquatic Conservation: Marine and Freshwater Ecosystems 34:e4055. https://doi.org/10.1002/aqc.4055

Hamilton S, Baker GB (2016) Current bycatch levels in Auckland Islands trawl fisheries unlikely to be driving New Zealand sea lion (*Phocarctos hookeri*) population decline. Aquatic Conservation: Marine and Freshwater Ecosystems 26:121–133. https://doi.org/10.1002/aqc.2524

Hamilton S, Baker GB, Browman H (2019) Population growth of an endangered pinniped - The New Zealand sea lion (*Phocarctos hookeri*) - Is limited more by high pup mortality than fisheries bycatch. ICES Journal of Marine Science 76:1794–1806. https://doi.org/10.1093/icesjms/fsz039

Hammond PS (2001) Marine mammals biology and conservation. In: Evans PGH, Raga JA (eds). Springer, Boston, MA

Hammond PS, Francis TB, Heinemann D, et al (2021) Estimating the Abundance of Marine Mammal Populations. Frontiers in Marine Science 8. https://doi.org/10.3389/fmars.2021.735770

Haney JC, Geiger HJ, Short JW (2014) Bird mortality from the Deepwater Horizon oil spill. II. Carcass sampling and exposure probability in the coastal Gulf of Mexico. Marine Ecology Progress Series513:239–252. https://doi.org/10.3354/meps10839

Harcourt RG (1992) Factors affecting early mortality in the South American fur seal (*Arctocephalus australis*) in Peru: density-related effects and predation. Journal of Zoology 226:259–270. https://doi.org/10.1111/j.1469-7998.1992.tb03838.x

Harcourt RG (2001) Advances in New Zealand mammalogy 1990–2000: Pinnipeds. Journal of the Royal Society of New Zealand 31:135–160. https://doi.org/10.1080/03014223.2001.9517644

Harcourt RG, Bradshaw CJA, Dickson K, Davis LS (2002) Foraging ecology of a generalist predator, the female New Zealand fur seal. Marine Ecology Progress Series 227:11–24. https://doi.org/10.3354/meps227011

Harte DS (2019) Evaluation of earthquake stochastic models based on their real-Time forecasts: A case study of Kaikoura 2016. Geophysical Journal International 217:1894–1914. https://doi.org/10.1093/gji/ggz088

Harvell CD, Mitchell CE, Ward JR, et al (2002) Climate Warming and Disease Risks for Terrestrial and Marine Biota. Science (1979) 296:2158–2162. https://doi.org/10.1126/science.1063699

Hatfield BB, Rathbun GB (1999) Interactions between northern elephant seals and vehicles near Point Piedras Blancas, California. Marine Mammal Science 15:598–600. https://doi.org/10.1111/j.1748-7692.1999.tb00829.x

Hauksson E (2007) Abundance of grey seals in Icelandic waters, based on trends of pup-counts from aerial surveys. NAMMCO Scientific Publications 6:85–97

Hay CH (2020) Seashore uplift and the distribution of the bull kelp *Durvillaea willana Lindauer* in New Zealand. New Zealand Journal of Botany 58:94–117. https://doi.org/10.1080/0028825X.2019.1679842

He K, Dai Q, Gu X, et al (2019) Effects of roads on giant panda distribution: a mountain range scale evaluation. Scientific Reports 9:1110. https://doi.org/10.1038/s41598-018-37447-0

Helm RC, Costa DP, DeBruyn TD, et al (2015) Overview of Effects of Oil Spills on Marine Mammals. In: Handbook of Oil Spill Science and Technology. Wiley, pp 455–475

Hendrix AM, Lefebvre KA, Quakenbush L, et al (2021) Ice seals as sentinels for algal toxin presence in the Pacific Arctic and subarctic marine ecosystems. Marine Mammal Science 37:1292–1308. https://doi.org/10.1111/mms.12822

Henkel LA, Ziccardi MH (2018) Life and death: How should we respond to oiled wildlife? Journal of Fish and Wildlife Management 9:296–301. https://doi.org/10.3996/062017-JFWM-054

Hernandez-Milian G, Lusher A, MacGabban S, Rogan E (2019) Microplastics in grey seal (*Halichoerus grypus*) intestines: Are they associated with parasite aggregations? Marine Pollution Bulletin 146:349–354. https://doi.org/10.1016/j.marpolbul.2019.06.014

Higham J (1998) Tourists and albatrosses: the dynamics of tourism at the Northern Royal Albatross Colony, Taiaroa Head, New Zealand. Tourism Management 19:521–531. https://doi.org/10.1016/S0261-5177(98)00054-5

Hindell MA, Burton HR (1988) The history of the elephant seal industry at Macquarie Island and an estimate of the pre-sealing numbers. Pap Proc 122:159–176. https://doi.org/10.26749/rstpp.122.1.159

Hobday AJ, Minstrell ML (2008) Distribution and abundance of roadkill on Tasmanian highways: Human management options. Wildlife Research 35:712–726. https://doi.org/10.1071/WR08067

Hoegh-Guldberg O, Mumby PJ, Hooten AJ, et al (2007) Coral Reefs Under Rapid Climate Change and Ocean Acidification. Science (1979) 318:1737–1742. https://doi.org/10.1126/science.1152509

Hoelzel AR (1999) Impact of population bottlenecks on genetic variation and the importance of life-history; a case study of the northern elephant seal. Biological Journal of the Linnean Society 68:23–39. https://doi.org/10.1111/j.1095-8312.1999.tb01156.x

Hong S, Yoon SJ, Kim T, et al (2020) Response to oiled wildlife in the management and evaluation of marine oil spills in South Korea: A review. Regional Studies in Marine Science 40:101542. https://doi.org/10.1016/j.rsma.2020.101542

Horswill C, Miller JAO, Wood MJ (2022) Impact assessments of wind farms on seabird populations that overlook existing drivers of demographic change should be treated with caution. Conservation Science and Practice 4: https://doi.org/10.1111/csp2.12644

Hoskins AJ, Schumann N, Costa DP, Arnould JPY (2017) Foraging niche separation in sympatric temperate-latitude fur seal species. Marine Ecology Progress Series 566:229–241. https://doi.org/10.3354/meps12024

Hradsky BA, Robley A, Alexander R, et al (2017) Human-modified habitats facilitate forest-dwelling populations of an invasive predator, *Vulpes vulpes*. Scientific Reports 7:12291. https://doi.org/10.1038/s41598-017-12464-7

Huang Y, Bird R, Bell M (2009) A comparative study of the emissions by road maintenance works and the disrupted traffic using life cycle assessment and micro-simulation. Transportation Research Part D: Transport and Environment 14:197–204. https://doi.org/10.1016/j.trd.2008.12.003

Hucke-Gaete R, Osman LP, Moreno CA, Torres D (2004) Examining natural population growth from near extinction: The case of the Antarctic fur seal at the South Shetlands, Antarctica. Polar Biology 27:304–311. https://doi.org/10.1007/s00300-003-0587-8

Hückstädt LA, McCarthy MD, Koch PL, Costa DP (2017) What difference does a century make? Shifts in the ecosystem structure of the Ross Sea, Antarctica, as evidenced from a sentinel species, the weddell seal. Proceedings of the Royal Society B: Biological Sciences 284: https://doi.org/10.1098/rspb.2017.0927

Hui TCY, Gryba R, Gregr EJ, Trites AW (2015) Assessment of competition between fisheries and steller sea lions in Alaska based on estimated prey biomass, fisheries removals and predator foraging behaviour. PLoS One 10:e0123786. https://doi.org/10.1371/journal.pone.0123786

Huijser MP, Begley JS (2022) Implementing wildlife fences along highways at the appropriate spatial scale: A case study of reducing road mortality of Florida Key deer. Nature Conservation 47:283–302. https://doi.org/10.3897/natureconservation.47.72321

Huijser MP, Fairbank ER, Camel-Means W, et al (2016) Effectiveness of short sections of wildlife fencing and crossing structures along highways in reducing wildlife-vehicle collisions and providing safe crossing opportunities for large mammals. Biological Conservation 197:61–68. https://doi.org/10.1016/j.biocon.2016.02.002

Hunter SA, Tennyson AJD, Bartle JA, et al (2019) Assessing avian mortality during oil spills: a case study of the New Zealand MV 'Rena' oil spill, 2011. Endangered Species Research 39:303–314. https://doi.org/10.3354/esr00974

Hupman K, Visser IN, Fyfe J, et al (2020) From Vagrant to Resident: occurrence, residency and births of leopard seals (*Hydrurga leptonyx*) in New Zealand waters. New Zealand Journal of Marine and Freshwater Research 54:1–23. https://doi.org/10.1080/00288330.2019.1619598

Hussain A, Fusté-Forné F (2021) Post-Pandemic Recovery: A Case of Domestic Tourism in Akaroa (South Island, New Zealand). World 2:127–138. https://doi.org/10.3390/world2010009

Inglis G, Gust N, Fitridge I, et al (2008) Port of Lyttelton Second baseline survey for non-indigenous marine species (Research Project ZBS2000-04) Prepared for MAFBNZ Post Border Directorate Disclaimer

IPIECA (2014) The global oil and gas industry association for environmental and social issues. IOGP Report 516.

Jackman JL, Vaske JJ, Dowling-Guyer S, et al (2024) Seals and the Marine Ecosystem: Attitudes, Ecological Benefits/Risks and Lethal Management Views. Human Dimensions of Wildlife 29:142–158. https://doi.org/10.1080/10871209.2023.2212686

Jaeger JAG, Fahrig L (2004) Effects of Road Fencing on Population Persistence. Conservation Biology 18:1651–1657. https://doi.org/10.1111/j.1523-1739.2004.00304.x

Jasińska KD, Żmihorski M, Krauze-Gryz D, et al (2019) Linking habitat composition, local population densities and traffic characteristics to spatial patterns of ungulate-train collisions. Journal of Applied Ecology 56:2630–2640. https://doi.org/10.1111/1365-2664.13495

Jepsen EM, de Bruyn PJN (2019) Pinniped entanglement in oceanic plastic pollution: A global review. Marine Pollution Bulletin 145:295–305. https://doi.org/10.1016/j.marpolbul.2019.05.042

Jernelv A (2010) The threats from oil spills: Now, then, and in the future. Ambio 39:353–366. https://doi.org/10.1007/s13280-010-0085-5

Jessup DA, Yeates LC, Sharon TC, et al (2012) Washing oiled sea otters. Wildlife Society Bulletin 36:6–15. https://doi.org/10.1002/wsb.113

Johnson GC, Lyman JM (2020) Warming trends increasingly dominate global ocean. Nature Climate Change 10:757–761. https://doi.org/10.1038/s41558-020-0822-0

Johnson WM, Lavigne DM (1999) Monk seals in antiquity. The Mediterranean monk seal 1–101.

Jolly CJ, Dickman CR, Doherty TS, et al (2022) Animal mortality during fire. Global Change Biology 28:2053–2065. https://doi.org/10.1111/gcb.16044

Karamanlidis A (2024) Using ‘pup multipliers’ to estimate demographic parameters of Mediterranean monk seals in the eastern Mediterranean Sea. Endangered Species Research 53:261–270. https://doi.org/10.3354/esr01301

Karanth KU, Nichols JD, Kumar NS, Hines JE (2006) Assessing tiger population dynamics using photographic capture-recapture sampling. Ecology 87:2925–2937. https://doi.org/10.1890/0012-9658(2006)87[2925:ATPDUP]2.0.CO;2

Kazemi VD, Jafari H, Yavari A (2016) Spatio-Temporal Patterns of Wildlife Road Mortality in Golestan National Park-North East of Iran. Open Journal of Ecology 06:312–324. https://doi.org/10.4236/oje.2016.66031

Keegan LJ, White RSA, Macinnis-Ng C (2022) New Zealand Ecological Society Current knowledge and potential impacts of climate change on New Zealand’s biological heritage. New Zealand Journal of Ecology 46:1–24. https://doi.org/10.2307/48652113

Keller BJ, Bender LC (2007) Bighorn Sheep Response to Road-Related Disturbances in Rocky Mountain National Park, Colorado. Journal of Wildlife Management 71:2329. https://doi.org/10.2193/2006-486

Kenyon KW, Wilke F (1953) Migration of the Northern Fur Seal, Callorhinus ursinus. Journal of Mammalogy 34:86–98. https://doi.org/https://doi.org/10.2307/1375947

Kim K, Yi Y, Woo D, et al (2021) Factors Influencing Roadkill Hotspot in the Republic of Korea. Proceedings of the National Institute of Ecology of the Republic of Korea 2:274–278. https://doi.org/10.22920/PNIE.2021.2.4.274

Kindsvater HK, Dulvy NK, Horswill C, et al (2018) Overcoming the Data Crisis in Biodiversity Conservation. Trends in Ecology & Evolution 33:676–688. https://doi.org/10.1016/j.tree.2018.06.004

King MD, Elliott JE, Williams TD (2021) Effects of petroleum exposure on birds: A review. Science of the Total Environment 755:142834. https://doi.org/10.1016/j.scitotenv.2020.142834

Kirkman SP, Lavigne DM (2010) Assessing the hunting practices of Namibia's commercial seal hunt. South African Journal of Science 106. https://doi.org/10.4102/sajs.v106i3/4.166

Kirkman SP, Oosthuizen W, Meÿer M, et al (2007) Making sense of censuses and dealing with missing data: trends in pup counts of Cape fur seal *Arctocephalus pusillus pusillus* for the period 1972–2004. African Journal of Marine Science 29:161–176. https://doi.org/10.2989/AJMS.2007.29.2.2.185

Kirkwood R, Pemberton D, Gales R, et al (2010) Continued population recovery by Australian fur seals. Marine and Freshwater Research 61:695–701. https://doi.org/10.1071/MF09213

Kiyota M (2021) Site Fidelity, Male Harassment and Female Gregariousness: Factors Shaping the Highly Polygynous Mating System of the Northern Fur Seal (*Callorhinus ursinus* (C. ursinus). In: Ethology and Behavioral Ecology of Otariids and the Odobenid. Springer, Cham., pp 161–182

Knape J, Jonzén N, SköLd M, et al (2011) Individual heterogeneity and senescence in Silvereyes on Heron Island. Ecology 92:813–820. https://doi.org/10.1890/10-0183.1

Knape J, Paquet M, Arlt D, et al (2023) Partitioning variance in population growth for models with environmental and demographic stochasticity. Journal of Animal Ecology 92:1979–1991. https://doi.org/10.1111/1365-2656.13990

Kobayashi Y, Kariya T, Chishima J, et al (2014) Population trends of the Kuril harbour seal *Phoca vitulina* stejnegeri from 1974 to 2010 in southeastern Hokkaido, Japan. Endangered Species Research 24:61–72. https://doi.org/10.3354/esr00553

Koemle D, Zinngrebe Y, Yu X (2018) Highway construction and wildlife populations: Evidence from Austria. Land use policy 73:447–457. https://doi.org/10.1016/j.landusepol.2018.02.021

Kovacs KM, Aguilar A, Aurioles D, et al (2012) Global threats to pinnipeds. Marine Mammal Science 28:414–436. https://doi.org/10.1111/j.1748-7692.2011.00479.x

Kučas A, Balčiauskas L (2021) Impact of road fencing on ungulate–vehicle collisions and hotspot patterns. Land (Basel) 10:338. https://doi.org/10.3390/land10040338

Kuhn CE, Baker JD, Towell RG, Ream RR (2014) Evidence of localized resource depletion following a natural colonization event by a large marine predator. Journal of Animal Ecology 83:1169–1177. https://doi.org/10.1111/1365-2656.12202

Kulp SA, Strauss BH (2019) New elevation data triple estimates of global vulnerability to sea-level rise and coastal flooding. Nature Communications 10:1–12. https://doi.org/10.1038/s41467-019-12808-z

Kunz BK, Little EE, Barandino VL (2022) Aquatic Toxicity of Chemical Road Dust Suppressants to Freshwater Organisms. Archives of Environmental Contamination and Toxicology 82:294–305. https://doi.org/10.1007/s00244-020-00806-y

Lai YC, Shieh B Sen, Kam YC (2007) Population patterns of a riparian frog (*Rana swinhoana*) before and after an earthquake in subtropical Taiwan. Biotropica 39:731–736. https://doi.org/10.1111/j.1744-7429.2007.00320.x

Laidre KL, Stern H, Kovacs KM, et al (2015) Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology 29:724–737. https://doi.org/10.1111/cobi.12474

Lalas C, Bradshaw CJA (2001) Folklore and chimerical numbers: Review of a millennium of interaction between fur seals and humans in the New Zealand region. New Zealand Journal of Marine and Freshwater Research35:477–497. https://doi.org/10.1080/00288330.2001.9517017

Lalas C, Harcourt RG (1995) Pup production of the New Zealand fur seal on Otago Peninsula, New Zealand. Journal of the Royal Society of New Zealand 25:81–88. https://doi.org/10.1080/03014223.1995.9517483

Lalas C, Murphy B (1998) Increase in the abundance of New Zealand fur seals at the Catlins, South Island, New Zealand. Journal of the Royal Society of New Zealand 28:287–294. https://doi.org/10.1080/03014223.1998.9517564

Lande R, Engen S, SÆther B-E (2003) Demographic and environmental stochasticity. In: Stochastic Population Dynamics in Ecology and Conservation. Oxford University Press, pp 1–24

Lander RH (1981) A life table and biomass estimate for Alaskan fur seals. Fisheries Research 1:55–70

Langen TA, Gunson KE, Scheiner CA, Boulerice JT (2012) Road mortality in freshwater turtles: Identifying causes of spatial patterns to optimize road planning and mitigation. Biodiversity and Conservation 21:3017–3034. https://doi.org/10.1007/s10531-012-0352-9

Lascelles B, Notarbartolo Di Sciara G, Agardy T, et al (2014) Migratory marine species: Their status, threats and conservation management needs. Aquatic Conservation: Marine and Freshwater Ecosystems 24:111–127. https://doi.org/10.1002/aqc.2512

Lau Y, Kanrak M, Ng AKY, Ling X (2023) Arctic region: analysis of cruise products, network structure, and popular routes. Polar Geography 46:157–169. https://doi.org/10.1080/1088937X.2023.2182381

Lea M-A, Hindell MarkA (1997) Pup Growth and Maternal Care in New Zealand Fur Seals, *Arctocephalus forsteri*, at Maatsuyker Island, Tasmania. Wildlife Research 24:307. https://doi.org/10.1071/WR96054

Lee DE, Berger RW, Tietz JR, et al (2018) Initial growth of northern fur seal (*Callorhinus ursinus*) colonies at the South Farallon, San Miguel, and Bogoslof Islands. Journal of Mammalogy 99:1529–1538. https://doi.org/10.1093/jmammal/gyy131

Lee T, Quinn MS, Duke D (2006) Citizen, Science, Highways, and Wildlife: Using a Web-based GIS to Engage Citizens in Collecting Wildlife Information. Ecology and Society 11:11. https://www.jstor.org/stable/26267782

Lefkovitch LP (1965) The Study of Population Growth in Organisms Grouped by Stages. Biometrics 1–18

Lehtonen E, Lehmonen R, Kostensalo J, et al (2022) Feasibility and effectiveness of seal deterrent in coastal trap-net fishing – development of a novel mobile deterrent. Fisheries Research 252. https://doi.org/10.1016/j.fishres.2022.106328

Leighton FA (1993) The toxicity of petroleum oils to birds. Environmental Reviews 1:92–103. https://doi.org/10.1139/a93-008

Leslie PH (1945) On the use of matrices in certain population mathematics. Biometrika 33:183–212. https://doi.org/10.2307/2332297

Levy S (2020) A Tale of Two Seal Hunts: Contesting the Conflation of Canadian Sealing Activities. Journal of International Wildlife Law and Policy 23:166–190. https://doi.org/10.1080/13880292.2020.1846858

Lewis JP, Tarnecki JH, Garner SB, et al (2020) Changes in Reef Fish Community Structure Following the Deepwater Horizon Oil Spill. Scientific Reports 10. https://doi.org/10.1038/s41598-020-62574-y

Lewis KB, Barnes PM (1999) Kaikoura Canyon, New Zealand: active conduit from near-shore sediment zones to trench-axis channel. Marine Geology 162:39–69. https://doi.org/https://doi.org/10.1016/S0025-3227(99)00075-4

Li X, Tzika AC, Liu Y, et al (2010) Preliminary genetic status of the spotted seal *Phoca largha* in Liaodong Bay (China) based on microsatellite and mitochondrial DNA analyses. Trends in Ecology & Evolution 2:6. https://doi.org/10.4081/eb.2010.e6

Lima M, Páez E (1995) Growth and Reproductive Patterns in the South American Fur Seal. Journal of Mammalogy 76:1249–1255. https://doi.org/10.2307/1382619

Lima M, Paez E (1997) Demography and population dynamics of South American fur seals. Journal of Mammalogy 78:914–920. https://doi.org/10.2307/1382951

Ling JK (1990) Exploitation of fur seals and sea lions from Australian, New Zealand and adjacent subantarctic islands during the eighteenth, nineteenth and twentieth centuries. Australian Zoologist 31:323–350. https://doi.org/https://doi.org/10.7882/AZ.1999.036

Lodé T (2000) Effect of a motorway on mortality and isolation of wildlife populations. Ambio 29:163–166. https://doi.org/10.1579/0044-7447-29.3.163

Lowry MS, Condit R, Hatfield B, et al (2014) Abundance, distribution, and population growth of the northern elephant seal (*Mirounga angustirostris*) in the United States from 1991 to 2010. Aquatic Mammals 40:20–31. https://doi.org/10.1578/AM.40.1.2014.20

Lunn NJ, Boyd IL (1991) Pupping-Site Fidelity of Antarctic Fur Seals at Bird Island, South Georgia. Journal of Mammalogy 72:202–206. https://doi.org/10.2307/1381999

Lynch M, Nielsen O, Duignan PJ, et al (2011) Serologic survey for potential pathogens and assessment of disease risk in Australian fur seals. Journal of Wildlife Diseases 47:555–565. https://doi.org/10.7589/0090-3558-47.3.555

Lyngdoh MK, Chettri A, Adhikari D, Barik SK (2018) Current Science Association Metapopulation modelling of threatened plants to assess conservation status and determine minimum viable population size. Current Science 114:532–538. https://www.jstor.org/stable/26495106

Lyons ET, Melin SR, Delong RL, et al (2001) Current prevalence of adult *Uncinaria* spp. in northern fur seal (*Callorhinus ursinus*) and California sea lion (*Zalophus californianus*) pups on San Miguel Island, California, with notes on the biology of these hookworms. International Journal of Wildland Fire 97:309–318. https://doi.org/10.1016/S0304-4017(01)00418-6

Lyttelton Port Company (2023) Focused on the Future Annual Report 2023. Retrieved from: https://www.lpc.co.nz/annual-report-2023

MacDiarmid A, Lalas C (2014) Rapid recolonisation of south-eastern New Zealand by New Zealand fur seals *Arctocephalus forsteri*. Wellington, New Zealand.

MacKenzie DI., Fletcher DJ., Dillingham PW., et al (2022) Updated spatially explicit fisheries risk assessment for New Zealand marine mammal populations. Fisheries New Zealand, Ministry for Primary Industries.

MacMillan H, Moore AB, Augé AA, Chilvers BL (2016) GIS-based multi-criteria analysis of breeding habitats for recolonising species: New Zealand sea lions. Ocean and Coastal Management 130:162–171. https://doi.org/10.1016/j.ocecoaman.2016.06.008

Majluf P (1992) Timing of births and juvenile mortality in the South American fur seal in Peru. Journal of Zoology 227:367–383. https://doi.org/10.1111/j.1469-7998.1992.tb04400.x

Malo JE, Suárez F, Díez A (2004) Can we mitigate animal-vehicle accidents using predictive models? Journal of Applied Ecology 41:701–710. https://doi.org/10.1111/j.0021-8901.2004.00929.x

Maloney A, Chilvers BL, Muller CG, Halcy M (2012) Increasing pup production of New Zealand sea lions at Campbell Island/Motu Ihupuku: Can it continue? New Zealand Journal of Zoology39:19–29. https://doi.org/10.1080/03014223.2011.614265

Manlik O, Lacy RC, Sherwin WB, et al (2022) A stochastic model for estimating sustainable limits to wildlife mortality in a changing world. Conservation Biology 36: https://doi.org/10.1111/cobi.13897

Manno K, Young M (2023) New Zealand sea lion/pakake/whakahao field research report. Wellington, New Zealand.

Maritime New Zealand (2021) New Zealand Marine Oil Spill Readiness and Response Strategy: 2022–2026. Wellington, New Zealand.

Marschall S, Granquist SM, Burns GL (2017) Interpretation in wildlife tourism: Assessing the effectiveness of signage on visitor behaviour at a seal watching site in Iceland. Journal of Outdoor Recreation and Tourism 17:11–19. https://doi.org/10.1016/j.jort.2016.11.001

Marsh DM, Milam GS, Gorham NP, Beckman NG (2005) Forest roads as partial barriers to terrestrial salamander movement. Conservation Biology 19:2004–2008. https://doi.org/10.1111/j.1523-1739.2005.00238.x

Massey C, Townsend D, Rathje E, et al (2018) Landslides Triggered by the 14 November 2016 Mw 7.8 Kaikōura Earthquake, New Zealand. Bulletin of the Seismological Society of America 108:1630–1648. https://doi.org/10.1785/0120170305

Masson-Delmotte V, Zhai P, Chen Y, et al (2021) Working Group I Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change

Mattern T, Wilson K-J (2018) New Zealand penguins-current knowledge and research priorities. A report complied for Birds New Zealand.

Mattlin RH (1978a) Population biology, thermoregulation and site preference of the New Zealand fur seal, *Arctocephalus forsteri* (lesson, 1828), on the Open Bay Islands, New Zealand. Doctoral Dissertation, University of Canterbury

Mattlin RH (1978b) Pup mortality of the New Zealand fur seal (*Arctocephalus forsteri* Lesson). New Zealand Journal of Ecology 1:138–144. https://www.jstor.org/stable/24052393

Mattlin RH, Gales NJ, Costa DP (1998) Seasonal dive behaviour of lactating New Zealand fur seals (*Arctocephalus forsteri*). Canadian Journal of Zoology 76:350–360. https://doi.org/10.1139/z97-187

Mbise FP, Fredriksen KE, Ranke PS, et al (2020) Human habituation reduces hyrax flight initiation distance in Serengeti. Ethology 126:297–303. https://doi.org/10.1111/eth.12968

Mcconkey S, Mcconnell H, Lalas C, et al (2002) A Northward Spread In The Breeding Distribution Of The New Zealand Sea Lion (Phocartos Hookeri). Australian Mammalogy 24:97–106. https://doi.org/10.1071/AM02097

McDonald L, Beauprez G, Gardner G, et al (2014) Range-wide population size of the lesser prairie-chicken: 2012 and 2013. Wildlife Society Bulletin 38:536–546. https://doi.org/10.1002/wsb.417

McIntosh RR, Kirkman SP, Thalmann S, et al (2018a) Understanding meta-population trends of the Australian fur seal, within sights for adaptive monitoring. PLoS One 13:e0200253. https://doi.org/10.1371/journal.pone.0200253

McIntosh RR, Holmberg R, Dann P (2018b) Looking without landing-using Remote Piloted Aircraft to monitor fur seal populations without disturbance. Frontiers in Marine Science 5. https://doi.org/10.3389/fmars.2018.00202

Mckenzie J (2006) Population demographics of New Zealand (*Arctocephalus Forsteri*). Doctoral Dissertation, La Trobe University

McKenzie J, Page B, Shaughnessy PD, Hindell MA (2007) Age and Reproductive Maturity of New Zealand Fur Seals (*Arctocephalus forsteri*) in Southern Australia. Journal of Mammalogy 88:639–648. https://doi.org/10.1644/06-MAMM-A-150R1.1

Mckenzie J, Parry LJ, Page B, Goldsworthy SD (2005) Estimation of pregnancy rates and reproductive failure in New Zealand fur seals (*Arctocephalus forsteri*). Journal of Mammalogy 86:1237–1246. https://doi.org/10.1644/05-MAMM-A-085R.1

McLean LJ, George S, Ierodiaconou D, et al (2018) Impact of rising sea levels on Australian fur seals. PeerJ e5786. https://doi.org/10.7717/peerj.5786

McNab AG, Crawley MC (1975) Mother and pup behaviour of the New Zealand fur seal, Arctocephalus forsteri (Lesson). 3:77–88

Mearns AJ, Levine E, Ender RY, et al (1999) Protecting fur seals during spill response: lessons from the San Jorge (Uruguay) Oil Spill. In: International Oil Spill Conference. American Petroleum Institute, pp 467–470

Meyer S, Robertson BC, Chilvers BL, Krkošek M (2015) Population dynamics reveal conservation priorities of the threatened New Zealand sea lion *Phocarctos hookeri*. Marine Biology 162:1587–1596. https://doi.org/10.1007/s00227-015-2695-8

Mikkelsen L, Hermannsen L, Beedholm K, et al (2017) Simulated seal scarer sounds scare porpoises, but not seals: Species-specific responses to 12 kHz deterrence sounds. Royal Society Open Science 4:170286. https://doi.org/10.1098/rsos.170286

Milano VN, Grandi MF, Schiavini ACM, Crespo EA (2020a) Recovery of South American fur seals from Fuegian Archipelago (Argentina). Marine Mammal Science 36:1022–1032. https://doi.org/10.1111/mms.12686

Milano VN, Grandi MF, Schiavini ACM, Crespo EA (2020b) Sea lions (*Otaria flavescens*) from the end of the world: insights of a recovery. Polar Biology 43:695–706. https://doi.org/10.1007/s00300-020-02672-9

Miller EH (1974) Social behaviour between adult male and female New Zealand fur seals, *Arctocephalus forsteri* (Lesson) during the breeding season. Australian Journal of Zoology 22:155. https://doi.org/10.1071/ZO9740155

Miller EH (1975) Annual cycle of fur seals, *Arctocephalus forsteri* (Lesson), on the Open Bay Islands, New Zealand

Moller TH, Molloy FC, Thomas HM (2003) Oil spill risks and the state of preparedness in the regional seas. In: International Oil Spill Conference, pp 919–922

Moore P, Moffat R (1990) Research and management projects on Campbell Island 1987-88.

Moore SE (2008) Marine mammals as ecosystem sentinels. Journal of Mammalogy 89:534–540. https://doi.org/https://doi.org/10.1644/07-MAMM-S-312R1.1

Morishita J (2008) What is the ecosystem approach for fisheries management? Marine Policy 32:19–26. https://doi.org/10.1016/j.marpol.2007.04.004

Mountjoy JJ, Howarth JD, Orpin AR, et al (2018) Earthquakes drive large-scale submarine canyon development and sediment supply to deep-ocean basins. Science Advances 4:eaar3748. https://doi.org/ 10.1126/sciadv.aar3748

Mullin J V., Champ MA (2003) Introduction/overview to in situ burning of oil spills. Spill Science & Technology Bulletin. 8:323–330. https://doi.org/10.1016/S1353-2561(03)00076-8

Muncaster SP, Jacobson G, Taiarui M, et al (2016) Effects of MV Rena heavy fuel oil and dispersed oil on yellowtail kingfish early life stages. New Zealand Journal of Marine and Freshwater Research 50:131–143. https://doi.org/10.1080/00288330.2015.1078821

Murphy KJ, Roberts DR, Jensen WF, et al (2023) Mule deer fawn recruitment dynamics in an energy disturbed landscape. Ecology and Evolution 13: https://doi.org/10.1002/ece3.9976

Navigatus Consulting (2015) Marine Oil Spill Risk Assessment 2015. Prepared by Navigatus Consulting for Maritime New Zealand. Auckland, New Zealand.

Nelli L, Langbein J, Watson P, Putman R (2018) Mapping risk: Quantifying and predicting the risk of deer-vehicle collisions on major roads in England. Mammalian Biology 91:71–78. https://doi.org/10.1016/j.mambio.2018.03.013

Nelson MA, Quakenbush LT, Taras BD, Committee IS (2019) Subsistence harvest of ringed, bearded, spotted, and ribbon seals in Alaska is sustainable. Endangered Species Research 40:1–16. https://doi.org/10.3354/ESR00973

New Zealand Transport Authority (2017) The Bulletin Kaikoura earthquake update (special edition). Retrieved from: www.nzta.govt.nz/projects/kaikoura-earthquake-response

Nijkamp H, Sessions S, Blanc P, Autret Y (2014) Arctic Oiled Wildlife Response: Exploring Potential and Limitations. In: International Oil Spill Conference Proceedings. American Petroleum Institute., pp 1569–1582

Noad MJ, Kniest E, Dunlop RA (2019) Boom to bust? Implications for the continued rapid growth of the eastern Australian humpback whale population despite recovery. Population Ecology 61:198–209. https://doi.org/10.1002/1438-390X.1014

Norris AL, Houser DS, Crocker DE (2010) Environment and activity affect skin temperature in breeding adult male elephant seals (Mirounga angustirostris). Journal of Experimental Biology 213:4205–4212. https://doi.org/10.1242/jeb.042135

Northrup JM, Pitt J, Muhly TB, et al (2012) Vehicle traffic shapes grizzly bear behaviour on a multiple-use landscape. Journal of Applied Ecology 49:1159–1167. https://doi.org/10.1111/j.1365-2664.2012.02180.x

Northwest Area Committee (2019) Section 9310 Northwest Wildlife Response Plan.

Nunny L, Simmonds MP, Butterworth A (2018) A review of seal killing practice in Europe: Implications for animal welfare. Marine Policy 98:121–132. https://doi.org/10.1016/j.marpol.2018.08.013

O'Donnell MS, Edmunds DR, Aldridge CL, et al (2019) Designing multi-scale hierarchical monitoring frameworks for wildlife to support management: a sage-grouse case study. Ecosphere 10. https://doi.org/10.1002/ecs2.2872

Okeke ES, Okoye CO, Chidike Ezeorba TP, et al (2022) Emerging bio-dispersant and bioremediation technologies as environmentally friendly management responses toward marine oil spill: A comprehensive review. Journal of Environmental Management 322:116123. https://doi.org/https://doi.org/10.1016/j.jenvman.2022.116123

Osiecka AN, Fearey J, Elwen S, Gridley T (2020) Prolonged Nursing in Cape fur Seals (*Arctocephalus pusillus pusillus*) at Cape Cross Colony, Namibia. African Zoology 55:233–239. https://doi.org/10.1080/15627020.2020.1768144

Özcan AU, Kutlutürk MM, Kuter S (2022) Modelling of road-kill hotspots in steppe landscape in Turkey. Landscape and Ecological Engineering 18:441–449. https://doi.org/10.1007/s11355-022-00515-9

Pacoureau N, Authier M, Delord K, et al (2017) Early-life density dependence effects on growth and survival in subantarctic fur seals. Population Ecology 59:139–155. https://doi.org/10.1007/s10144-017-0573-6

Page B, McKenzie J, Goldsworthy S (2005) Inter-sexual differences in New Zealand fur seal diving behaviour. Marine Ecology Progress Series304:249–264. https://doi.org/10.3354/meps304249

Page B, McKenzie J, Sumner MD, et al (2006) Spatial separation of foraging habitats among New Zealand fur seals. Marine Ecology Progress Series. https://doi.org/doi:10.3354/meps323263

Paine RT, Ruesink JL, Sun A, et al (1996) Trouble on oiled waters: Lessons from the Exxon valdez oil spill. Annual Review of Ecology, Evolution, and Systematics 27:197–235. https://doi.org/10.1146/annurev.ecolsys.27.1.197

Pasamontes A, Aksenov AA, Schivo M, et al (2017) Noninvasive Respiratory Metabolite Analysis Associated with Clinical Disease in Cetaceans: A Deepwater Horizon Oil Spill Study. Environmental Science & Technology 51:5737–5746. https://doi.org/10.1021/acs.est.6b06482

Pavanato H, Schattschneider J, Childerhouse S, Briscoe D (2023a) Assessment of New. Zealand fur seal /kekeno bycatch by trawlers in the Cook Strait hoki fishery. Report prepared by Cawthron for Department of Conservation

Pavanato H, Schattschneider J, Childerhouse S, Briscoe D (2023b) Fur seal population estimate and bycatch analysis, Cook Strait hoki fishery. POP2021-06 final report prepared by Cawthron for Department of Conservation

Pavez G, Muñoz L, Inostroza P, Sepúlveda M (2011) Behavioral response of South American sea lion *Otaria flavescens* to tourist disturbance during the breeding season. Revista de Biología Marina y Oceanografía 46:135–140. https://doi.org/10.4067/S0718-19572011000200004

Payne MR (1977) Growth of a fur seal population. Philosophical Transactions of the Royal Society of London B, Biological Sciences 279:67–79. https://doi.org/10.1098/rstb.1977.0072

Perez-Venegas DJ, Valenzuela-Sánchez A, Montalva F, et al (2021) Towards understanding the effects of oceanic plastic pollution on population growth for a South American fur seal (*Arctocephalus australis australis*) colony in Chile. Environmental Pollution 279. https://doi.org/10.1016/j.envpol.2021.116881

Phillips A, Mathews E (2007) Comparison of survey methods for estimating abundance of harbor seal (*Phoca vitulina*) in glacial fiords. Fishery Bulletin 105:348–356

Plaza PI, Gamarra-Toledo V, Euguí JR, Lambertucci SA (2024) Recent Changes in Patterns of Mammal Infection with Highly Pathogenic Avian Influenza A(H5N1) Virus Worldwide. Emerging Infectious Diseases 30. https://doi.org/10.3201/eid3003.231098

Prince RC (2015) Oil spill dispersants: Boon or bane? Environmental Science & Technology 49:6376–6384. https://doi.org/10.1021/acs.est.5b00961

Pulit-Penaloza JA, Brock N, Belser JA, et al (2024) Highly pathogenic avian influenza A(H5N1) virus of clade 2.3.4.4b isolated from a human case in Chile causes fatal disease and transmits between co-housed ferrets. Emerging Microbes & Infections. https://doi.org/10.1080/22221751.2024.2332667

Punt AE, Butterworth DS (1995) The effects of future consumption by the Cape fur seal on catches and catch rates of the cape hakes. 4. modelling the biological interaction between Cape fur seals *Arctocephalus pusillus pusillus* and the Cape hakes *Merluccius capensis* and *M. paradoxus*. South African Journal of Marine Science 16:255–285. https://doi.org/10.2989/025776195784156494

Puryear W, Sawatzki K, Hill N, et al (2023) Highly Pathogenic Avian Influenza A(H5N1) Virus Outbreak in New England Seals, United States. Emerging Infectious Diseases 29:786–791. https://doi.org/10.3201/eid2904.221538

Queirolo D, Ahumada M, Martín MAS, et al (2025) Testing and implementation of an exclusion device for mitigating the incidental capture of sea lions in the Chilean small-scale trawl hake fishery. Fisheries Research 281:107205. https://doi.org/10.1016/j.fishres.2024.107205

R Development Core Team (2022) R: A Language and Environment for Statistical Computing. https://www.r-project.org/

Radio New Zealand (2023a) Grounded Austro Carina trawler at Banks Peninsula starting to break up, salvage could take months. Retrieved from: https://www.rnz.co.nz/news/national/499500/grounded-austro-carina-trawler-at-banks-peninsula-starting-to-break-up-salvage-could-take-months.

Radio New Zealand (2023b) Oil spill reported off coast of Banks Peninsula. Retrieved from: https://www.rnz.co.nz/news/national/498755/oil-spill-reported-off-coast-of-banks-peninsula

Raithel JD, Kauffman MJ, Pletscher DH (2007) Impact of Spatial and Temporal Variation in Calf Survival on the Growth of Elk Populations. Journal of Wildlife Management 71:795–803. https://doi.org/10.2193/2005-608

Ramp D, Ben-Ami D (2006) The Effect of Road-Based Fatalities on the Viability of a Peri-Urban Swamp Wallaby Population. Journal of Wildlife Management 70:1615–1624. https://doi.org/10.2193/0022-541x(2006)70[1615:teorfo]2.0.co;2

Raymond S, Schwartz ALW, Thomas RJ, et al (2021) Temporal patterns of wildlife roadkill in the UK. PLoS One 16:e0258083. https://doi.org/10.1371/journal.pone.0258083

Reeves RR, McClellan K, Werner TB (2013) Marine mammal bycatch in gillnet and other entangling net fisheries, 1990 to 2011. Endangered Species Research 20:71–97. https://doi.org/10.3354/esr00481

Reichmuth C, Holt MM, Mulsow J, et al (2013) Comparative assessment of amphibious hearing in pinnipeds. Journal of Comparative Physiology A 199:491–507. https://doi.org/10.1007/s00359-013-0813-y

Reid K, Forcada J (2005) Causes of offspring mortality in the Antarctic fur seal, *Arctocephalus gazella*: The interaction of density dependence and ecosystem variability. Canadian Journal of Zoology 83:604–609. https://doi.org/10.1139/Z05-045

Reiley BM, Benson TJ, Bednarz JC (2013) Mechanisms of Flood-Induced Territory Abandonment in an Obligate Ground-Foraging Bird. Condor 115:650–658. https://doi.org/10.1525/cond.2013.120163

Rendall AR, Webb V, Sutherland DR, et al (2021) Where wildlife and traffic collide: Roadkill rates change through time in a wildlife-tourism hotspot. Global Ecology and Conservation 27:e01530. https://doi.org/10.1016/j.gecco.2021.e01530

Rick TC, DeLong RL, Erlandson JM, et al (2011) Where were the northern elephant seals? holocene archaeology and biogeography of *mirounga angustirostris*. Holocene 21:1159–1166. https://doi.org/10.1177/0959683611400463

Riffaut L, McCoy K, Tirard C, et al (2005) Population genetics of the common guillemot, *Uria aalge*, in the North Atlantic: geographic impact of oil spills. Marine Ecology Progress Series 291:263–273. https://doi.org/10.3354/meps291263

Roberts J, Neale D (2016) Census & individual size of New Zealand fur seal/kekeno pups on the West Coast South Island from 1991 to 2016. Prepared for Department of Conservation. Wellington

Robertson BC, Gemmell NJ (2005) Microsatellite DNA markers for the study of population structure in the New Zealand fur seal Arctocephalus forsteri. Department of Conservation.

Robinson LM, Elith J, Hobday AJ, et al (2011) Pushing the limits in marine species distribution modelling: lessons from the land present challenges and opportunities. Global Ecology and Biogeography 20:789–802. https://doi.org/10.1111/j.1466-8238.2010.00636.x

Robinson RA, Morrison CA, Baillie SR (2014) Integrating demographic data: Towards a framework for monitoring wildlife populations at large spatial scales. Methods in Ecology and Evolution 5:1361–1372. https://doi.org/10.1111/2041-210X.12204

Robinson SJ, Harting AL, Mercer T, et al (2021) Sighting patterns reveal unobserved pupping events to revise reproductive rate estimates for Hawaiian monk seals in the main Hawaiian Islands. Marine Mammal Science 37:420–432. https://doi.org/10.1111/mms.12761

Rochman CM (2015) The complex mixture, fate and toxicity of chemicals associated with plastic debris in the marine environment. In: Bergmann M, Gutow L, Klages M (eds) Marine anthropogenic litter. Springer, Cham., pp 117–140

Rodríguez-Lozano P, Verkaik I, Rieradevall M, Prat N (2015) Small but powerful: Top predator local extinction affects ecosystem structure and function in an intermittent stream. PLoS One 10:ee0117630. https://doi.org/10.1371/journal.pone.0117630

Rogers TL (2009) Leopard seal: *Hydrurga leptonyx*. In: Encyclopedia of marine mammals. Academic Press, pp 673–674.

Rolandsen CM, Solberg EJ, Herfindal I, et al (2011) Large-scale spatiotemporal variation in road mortality of moose: Is it all about population density? Ecosphere 2:1–12. https://doi.org/10.1890/es11-00169.1

Rossi SP, Cox SP, Hammill MO, et al (2021) Forecasting the response of a recovered pinniped population to sustainable harvest strategies that reduce their impact as predators. ICES Journal of Marine Science 78:1804–1814. https://doi.org/10.1093/icesjms/fsa

Roux JP (1987) Recolonisation processes in the subantarctic fur seal, Arctocephalus tropicalis, on Amsterdam Island. In: Croxall JP, Gentry RL (eds) Status, biology, and ecology of fur seals. NOAA Technical Report., pp 189–194

Russell DJF, Morris CD, Duck CD, et al (2019) Monitoring long-term changes in UK grey seal pup production. Aquatic Conservation: Marine and Freshwater Ecosystems 29:24–39. https://doi.org/10.1002/aqc.3100

Ryan CJ, Hickling GJ, Wilson KJ (1997) Breeding habitat preferences of the New Zealand fur seal (*Arctocephalus forsteri*) on Banks Peninsula. Wildlife Research 24:225–235. https://doi.org/10.1071/WR95068

Rytwinski T, Soanes K, Jaeger JAG, et al (2016) How effective is road mitigation at reducing road-kill? A meta-analysis. PLoS One 11:e0166941. https://doi.org/10.1371/journal.pone.0166941

Saab ME, Vanier G, Sudlovenick E, et al (2023) Occurrence and antimicrobial resistance of Salmonella species and potentially pathogenic Escherichia coli in free-living seals of Canadian Atlantic and eastern Arctic waters. Zoonoses Public Health 70:542–554. https://doi.org/10.1111/zph.13064

Safford TG, Ulrich JD, Hamilton LC (2012) Public perceptions of the response to the Deepwater Horizon oil spill: Personal experiences, information sources, and social context. Journal of Environmental Management 113:31–39. https://doi.org/10.1016/j.jenvman.2012.08.022

Sagnol O, Richter C, Reitsma F, Field L (2015) Estimating sperm whale (*Physeter macrocephalus*) daily abundance from a shore-based survey within the Kaikoura submarine canyon, New Zealand. New Zealand Journal of Marine and Freshwater Research 49:41–50. https://doi.org/10.1080/00288330.2014.925479

Saltz D, Berger-Tal O, Motro U, et al (2019) Conservation implications of habituation in Nubian ibex in response to ecotourism. Animal Conservation 22:220–227. https://doi.org/10.1111/acv.12456

Santos RAL, Mota-Ferreira M, Aguiar LMS, Ascensão F (2018) Predicting wildlife road-crossing probability from roadkill data using occupancy-detection models. Science of the Total Environment 642:629–637. https://doi.org/10.1016/j.scitotenv.2018.06.107

Sawyer H, Lebeau C, Hart T (2012) Mitigating roadway impacts to migratory mule deer-A case study with underpasses and continuous fencing. Wildlife Society Bulletin 36:492–498. https://doi.org/10.1002/wsb.166

Schalk CM, Lutterschmidt WI, Weidler JM (2019) Hot moments and hot spots in the bayou: Spatiotemporal patterns of road occurrence in a snake assemblage in Louisiana, USA. Herpetological Conservation and Biology 14:533–545

Schaub M, Kéry M (2022) Introduction to Stage-Structured Population Models. In: Integrated Population Models. Elsevier, pp 67–115

Schiel DR, Alestra T, Gerrity S, et al (2019) The Kaikōura earthquake in southern New Zealand: Loss of connectivity of marine communities and the necessity of a cross-ecosystem perspective. Aquatic Conservation: Marine and Freshwater Ecosystems 29:1520–1534. https://doi.org/10.1002/aqc.3122

Schiel DR, Ross PM, Battershill CN (2016) Environmental effects of the MV Rena shipwreck: cross-disciplinary investigations of oil and debris impacts on a coastal ecosystem. New Zealand Journal of Marine and Freshwater Research 50:1–9. https://doi.org/10.1080/00288330.2015.1133665

Schultz JK, Marshall AJ, Pfunder M (2010) Genome-wide loss of diversity in the critically endangered Hawaiian monk seal. Diversity (Basel) 2:863–880. https://doi.org/10.3390/d2060863

Schulz TM, Bowen WD (2004) Pinniped lactation strategies: Evaluation of data on maternal and offspring life history traits. Marine Mammal Science 20:86–114. https://doi.org/10.1111/j.1748-7692.2004.tb01142.x

Schunck F, Melo MA, Matinata BS (2023) Extra-marine occurrence of the Magnificent Frigatebird *Fregata magnificens* in the Metropolitan Region of São Paulo, Southeast Brazil. Marine Ornithology 51:

Schwartz ALW, Williams HF, Chadwick E, et al (2018) Roadkill scavenging behaviour in an urban environment. Journal of Urban Ecology 4:juy006. https://doi.org/10.1093/jue/juy006

Schwing PT, Montagna PA, Joye SB, et al (2020) A Synthesis of Deep Benthic Faunal Impacts and Resilience Following the Deepwater Horizon Oil Spill. Frontiers in Marine Science 7:560012. https://doi.org/10.3389/fmars.2020.560012

Seber GAF (1982) The Estimation of Animal Abundance and Related Parameters. MacMillan Press, New York

Secco H, da Costa VO, Guerreiro M, Gonçalves PR (2022) Evaluating impacts of road expansion on porcupines in a biodiversity hotspot. Transportation Research Part D: Transport and Environment 102:103151. https://doi.org/10.1016/j.trd.2021.103151

Seiler A (2003) The toll of the automobile: wildlife and roads in Sweden. Doctoral Dissertation, Swedish University of Agricultural Sciences

Seiler A (2005) Predicting locations of moose-vehicle collisions in Sweden. Journal of Applied Ecology 42:371–382. https://doi.org/10.1111/j.1365-2664.2005.01013.x

Sella KAN, Ware M, Ceriani SA, et al (2023) Urban pocket beaches as nesting habitat for marine turtles: Their importance and risk from inundation. Global Ecology and Conservation 41:e02366. https://doi.org/10.1016/j.gecco.2023.e02366

Sepúlveda M, Quiñones RA, Esparza C, et al (2020) Vulnerability of a top marine predator to coastal storms: a relationship between hydrodynamic drivers and stranding rates of newborn pinnipeds. Scientific Reports 10. https://doi.org/10.1038/s41598-020-69124-6

Shaughnessy PD, Bossley M, Nicholls AO (2018) Fur seals and sea lions (family Otariidae) on the breakwaters at Adelaide's Outer Harbor, South Australia. Australian Mammalogy 40:157–161. https://doi.org/10.1071/AM17001

Shaughnessy PD, Gales NJ, Dennis TE, Goldsworthy SD (1994) Distribution and abundance of New Zealand fur seals, *Arctocephalus forsteri*, in South Australia and Western Australia. Wildlife Research 21:667. https://doi.org/10.1071/WR9940667

Shaughnessy PD, Goldsworthy SD (2015) Increasing abundance of pups of the long-nosed fur seal (*Arctocephalus forsteri*) on Kangaroo Island, South Australia, over 26 breeding seasons to 2013-14. Wildlife Research 42:619–632. https://doi.org/10.1071/WR14209

Shaughnessy PD, Goldsworthy SD, Libke JA (1995) Changes in the Abundance of New Zealand Fur Seals, *Arctocephalus forsteri*, on Kangaroo Island, South Australia. Wildlife Research 22:201–216. https://doi.org/https://doi.org/10.1071/WR9950201

Shaughnessy PD, Goldsworthy SD, Mackay AI (2014) Status and trends in abundance of New Zealand fur seal populations in South Australia Final Report to the Australian Marine Mammal Centre

Shaughnessy PD, Goldsworthy SD, Mackay AI (2015) The long-nosed fur seal (*Arctocephalus forsteri*) in South Australia in 2013-14: Abundance, status and trends. Australian Journal of Zoology 63:101–110. https://doi.org/10.1071/ZO14103

Sievwright KA (2014) Post-release survival and productivity of oiled little blue penguins (*Eudyptula minor*) rehabilitated after the 2011 C/V Rena oil spill. Masters Dissertation, Massey University

Sievwright KA, Battley PF, McConnell H, et al (2019) Survival rates of oil-rehabilitated and non-rehabilitated little penguins after the C/V Rena oil spill, New Zealand. Marine Pollution Bulletin 146:317–325. https://doi.org/10.1016/j.marpolbul.2019.06.027

Simmons DG, Fairweather JR (1998) Towards a Tourism Plan for Kaikōura. Tourism Research and Education Centre: Lincoln University. Lincoln, New Zealand.

Simonis AE, Forney KA, Rankin S, et al (2020) Seal Bomb Noise as a Potential Threat to Monterey Bay Harbor Porpoise. Frontiers in Marine Science 7. https://doi.org/10.3389/fmars.2020.00142

Soares CM, Dias RI (2020) Look both ways: factors affecting roadkill probability in Blue-black Grassquits (*Volatinia jacarina*). Canadian Journal of Zoology 98:603–610. https://doi.org/10.1139/cjz-2020-0004

Song J-S, Lee K-J, Ki K-S, Jun I-Y (2011) The Efficiency and Improvement of the Highway Wildlife Fences for Decrease of Mammals Roadkill 1a-In Case of Manjong~Hongchun Section on Jungang Highway. Korean Journal of Environment and Ecology 25:649–657

Soofi M, Qashqaei AT, Trei JN, et al (2022) A novel application of hierarchical modelling to decouple sampling artifacts from socio-ecological effects on poaching intensity. Biological Conservation 267:109488. https://doi.org/10.1016/j.biocon.2022.109488

Sorensen JH (1969) New Zealand fur seals with special reference to the 1946 open season. Fisheries Technical Report No. 42. Wellington

Sorrell KJ, Clarke RH, Holmberg R, McIntosh RR (2019) Remotely piloted aircraft improve precision of capture–mark–resight population estimates of Australian fur seals. Ecosphere 10: https://doi.org/10.1002/ecs2.2812

Soto KH, Trites AW, Arias-Schreiber M (2004) The effects of prey availability on pup mortality and the timing of birth of South American sea lions (*Otaria flavescens*) in Peru. Journal of Zoology 264:419–428. https://doi.org/10.1017/S0952836904005965

Southwell D, Wilkinson D, Hao T, et al (2022) A gap analysis of reconnaissance surveys assessing the impact of the 2019–20 wildfires on vertebrates in Australia. Biological Conservation 270. https://doi.org/10.1016/j.biocon.2022.109573

Soykan CU, Moore JE, Ždelis R, et al (2008) Why study bycatch? An introduction to the Theme Section on fisheries bycatch. Endangered Species Research 5:91–102. https://doi.org/10.3354/esr00175

Spanowicz AG, Teixeira FZ, Jaeger JAG (2020) An adaptive plan for prioritizing road sections for fencing to reduce animal mortality. Conservation Biology 34:1210–1220. https://doi.org/10.1111/cobi.13502

Stafford-Bell R, Scarr M, Scarpaci C (2012) Behavioural responses of the Australian Fur Seal (*Arctocephalus pusillus doriferus*) to vessel traffic and presence of swimmers in Port Phillip Bay, Victoria, Australia. Aquatic Mammals 38:241–249. https://doi.org/10.1578/AM.38.3.2012.241

Stenson GB (2014) Updated Estimates of Harp Seal Removals in the Northwest Atlantic. Canadian Science Advisory Secretariat.

Stewart EJ, Draper D (2008) The Sinking of the MS Explorer: Implications for Cruise Tourism in Arctic. Arctic 61:224–228

Stirling I (1971) Studies on the behaviour on the South Ausralian fur seal, *Arctocephalus forsteri* (Lesson) II. Adult females and pups. Australian Journal of Zoology 19:267. https://doi.org/10.1071/ZO9710267

Stoffel MA, Humble E, Paijmans AJ, et al (2018) Demographic histories and genetic diversity across pinnipeds are shaped by human exploitation, ecology and life-history. Nature Communications 9. https://doi.org/10.1038/s41467-018-06695-z

Stonehouse B (1965) Marine birds and mammals at Kaikoura. Proceedings (New Zealand Ecological Society) 12:13–20

Street R (1964) Feeding habits of the New Zealand fur seal (*Arctocephalus forsteri*). Wellington, New Zealand.

Stringell TB, Millar CP, Sanderson WG, et al (2014) When aerial surveys will not do: Grey seal pup production in cryptic habitats of Wales. In: Journal of the Marine Biological Association of the United Kingdom. Cambridge University Press, pp 1155–1159.

Stringer J, Brook MS, Justice R (2021) Post-earthquake monitoring of landslides along the Southern Kaikōura Transport Corridor, New Zealand. Landslides 18:409–423. https://doi.org/10.1007/s10346-020-01543-y

Stubben C (2020) stoch.projection: Project stochastic growth from a sequence of matrices. Retrieved from: https://search.r-project.org/CRAN/refmans/popbio/html/stoch.projection.html

Stubben C, Milligan B (2007) Journal of Statistical Software Estimating and Analyzing Demographic Models Using the popbio Package in R. Journal of Statistical Software 22: 1–23. https://doi.org/10.18637/jss.v022.i11

Suuronen P, Lunneryd SG, Königson S, et al (2023) Reassessing the management criteria of growing seal populations: The case of Baltic grey seal and coastal fishery. Marine Policy 155. https://doi.org/10.1016/j.marpol.2023.105684

Tanasovici RM, Kitahara M V., Dias GM (2020) Invasive coral *Tubastraea* spp. population growth in artificial habitats and its consequences to the diversity of benthic organisms. Marine Biology 167:119. https://doi.org/10.1007/s00227-020-03734-6

Taylor BD, Goldingay RL (2010) Roads and wildlife: Impacts, mitigation and implications for wildlife management in Australia. Wildlife Research 37:320–331. https://doi.org/10.1071/WR09171

Taylor JC, Bates SB, Whiting JC, et al (2022) Using camera traps to estimate ungulate abundance: a comparison of mark–resight methods. Remote Sensing in Ecology and Conservation 8:32–44. https://doi.org/10.1002/rse2.226

Taylor RH (1982) New Zealand fur seals at the Bounty Islands. New Zealand Journal of Marine and Freshwater Research16:1–9. https://doi.org/10.1080/00288330.1982.9515942

Taylor RH (1992) New Zealand fur seals at the Antipodes Islands. Journal of the Royal Society of New Zealand 22:107–122. https://doi.org/10.1080/03036758.1992.10420809

Taylor RH (1996) Distribution, abundance and pup production of the New Zealand fur seal (*Arctocephalus foresteri* Lesson) at the Bounty Islands (Vol. 32). Department of Conservation, Wellington

Taylor RH, Barton KJ, Wilson PR, et al (1995) Population status and breeding of New Zealand fur seals (*Arctocephalus forsteri*) in the Nelson-northern Marlborough region, 1991–94. New Zealand Journal of Marine and Freshwater Research 29:223–234. https://doi.org/10.1080/00288330.1995.9516656

Taylor S, Lynch M, Terkildsen M, et al (2018) Utility of fur as a biomarker for persistent organic pollutants in Australian fur seals (*Arctocephalus pusillus doriferus*). Science of The Total Environment 610–611:1310–1320. https://doi.org/10.1016/j.scitotenv.2017.08.204

Thomas L, Russell DJF, Duck CD, et al (2019) Modelling the population size and dynamics of the British grey seal. Aquatic Conservation: Marine and Freshwater Ecosystems 29:6–23. https://doi.org/10.1002/aqc.3134

Thomson RB, Butterworth DS, Boyd IL, Croxall JP (2000) Modeling the consequences of Antarctic krill harvesting on antarctic fur seals. Ecological Applications 10:1806–1819. https://doi.org/10.1890/1051-0761(2000)010[1806:MTCOAK]2.0.CO;2

Tidwell KS, Carrothers BA, Blumstein DT, Schakner ZA (2021) Steller Sea Lion (*Eumetopias jubatus*) Response to Non-lethal Hazing at Bonneville Dam. Frontiers in Conservation Science 2. https://doi.org/10.3389/fcosc.2021.760866

Torney CJ, Lloyd-Jones DJ, Chevallier M, et al (2019) A comparison of deep learning and citizen science techniques for counting wildlife in aerial survey images. Methods in Ecology and Evolution 10:779–787. https://doi.org/10.1111/2041-210X.13165

Toulec T, Lhota S, Soumarová H, et al (2020) Shrimp farms, fire or palm oil? Changing causes of proboscis monkey habitat loss. Global Ecology and Conservation 21:e00863. https://doi.org/10.1016/j.gecco.2019.e00863

Trites AW, Larkin PA (1996) Changes in the abundance of Steller sea lions (*Eumetopias jubatus*) in Alaska from 1956 to 1992: how many were there? Aquatic Mammals 22:153–166.

Túnez JI, Cappozzo HL, Cassini MH (2008) Natural and anthropogenic factors associated with the distribution of South American sea lion along the Atlantic coast. Hydrobiologia 598:191–202. https://doi.org/10.1007/s10750-007-9150-x

Ulloa M, Fernández A, Ariyama N, et al (2023) Mass mortality event in South American sea lions (*Otaria flavescens*) correlated to highly pathogenic avian influenza (HPAI) H5N1 outbreak in Chile. Veterinary Quarterly 43:1–10. https://doi.org/10.1080/01652176.2023.2265173

Ureña-Aranda CA, Rojas-Soto O, Martínez-Meyer E, et al (2015) Using range-wide abundance modeling to identify key conservation areas for the micro-endemic Bolson tortoise (*Gopherus flavomarginatus*). PLoS One 10:e0131452. https://doi.org/10.1371/journal.pone.0131452

Ventikos NP, Vergetis E, Psaraftis HN, Triantafyllou G (2004) A high-level synthesis of oil spill response equipment and countermeasures. The Journal of Hazardous Materials 107:51–58. https://doi.org/10.1016/j.jhazmat.2003.11.009

Vetemaa M, Päädam U, Fjälling A, et al (2021) Seal-induced losses and successful mitigation using acoustic harassment devices in estonian baltic trap-net fisheries. Proceedings of the Estonian Academy of Sciences 70:207–214. https://doi.org/10.3176/PROC.2021.2.09

Vincent C, Huon M, Caurant F, et al (2017) Grey and harbour seals in France: Distribution at sea, connectivity and trends in abundance at haulout sites. Deep Sea Research Part II: Topical Studies in Oceanography 141:294–305. https://doi.org/10.1016/j.dsr2.2017.04.004

Voyles J, Kilpatrick AM, Collins JP, et al (2014) Moving Beyond Too Little, Too Late: Managing Emerging Infectious Diseases in Wild Populations Requires International Policy and Partnerships. Ecohealth 12:404–407. https://doi.org/10.1007/s10393-014-0980-5

Vyas R, Vasava A (2019) Mugger Crocodile (*Crocodylus palustris*) Mortality due to Roads and Railways in Gujarat, India. Herpetological Conservation and Biology 14:615–626

Wang Y, Yang Y, Han Y, et al (2022) Temporal patterns and factors influencing vertebrate roadkill in China. Transportation Research Interdisciplinary Perspectives 15:100662. https://doi.org/10.1016/j.trip.2022.100662

Watson DM, Beaven B, Bradshaw CJA (2015) Population trends of New Zealand fur seals in the Rakiura region based on long-term population surveys and traditional ecological knowledge. New Zealand Journal of Marine and Freshwater Research 49:106–118. https://doi.org/10.1080/00288330.2014.953184

Watson DM, Lalas C, Seddon PJ (2009) Calibrations to estimate absolute numbers of New Zealand fur seal (*Arctocephalus forsteri*) pups from direct counts. New Zealand Journal of Marine and Freshwater Research 43:1053–1060. https://doi.org/10.1080/00288330.2009.9626528

Wege M, Etienne MP, Chris Oosthuizen W, et al (2016) Trend changes in sympatric Subantarctic and Antarctic fur seal pup populations at Marion Island, Southern Ocean. Marine Mammal Science 32:960–982. https://doi.org/10.1111/mms.12306

Wege M, Nevoux M, De Bruyn PJN, Bester MN (2014) Multi-state mark-recapture models as a novel approach to estimate factors affecting attendance patterns of lactating subantarctic fur seals from Marion Island. Antarctic Science 30:252–262. https://doi.org/10.1017/S0954102014000716

Weinberger CS, Vianna JA, Faugeron S, Marquet PA (2021) Inferring the impact of past climate changes and hunting on the South American sea lion. Diversity and Distributions 27:2479–2497. https://doi.org/10.1111/ddi.13421

Whitmer ER, Elias BA, Harvey DJ, Ziccardi MH (2018) An experimental study of the effects of chemically dispersed oil on feather structure and waterproofing in Common Murres (*Uria aalge*). Journal of Wildlife Diseases 54:315–328. https://doi.org/10.7589/2017-01-016

Whittington J, Low P, Hunt B (2019) Temporal road closures improve habitat quality for wildlife. Scientific Reports 9:3772. https://doi.org/10.1038/s41598-019-40581-y

Wickens P, York AE (1997) Comparative population dynamics of fur seals. Marine Mammal Science 13:241–292. https://doi.org/10.1111/j.1748-7692.1997.tb00631.x

Williams JC, Drummond BA, Buxton RT (2010) Initial effects of the August 2008 volcanic eruption on breeding birds and marine mammals at Kasatochi Island, Alaska. Arctic, Antarctic, and Alpine Research 42:306–314. https://doi.org/10.1657/1938-4246-42.3.306

Williams R, Gero S, Bejder L, et al (2011) Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the Deepwater Horizon/BP incident. Conservation Letters 4:228–233. https://doi.org/10.1111/j.1755-263X.2011.00168.x

Wilson G (1981) Distribution and abundance of the New Zealand fur seal, *Arctocephalus forsteri*. Wellington, New Zealand.

Wilson J, Shone MC, Simmons DG, Stewart E (2015a) Making waves: Examining the interface between cruise tourism and destination community in Akaroa, New Zealand. Tourism in Marine Environments 10:211–223. https://doi.org/10.3727/154427315X14181438892856

Wilson N, Prickett M, Boyd M (2023) Food security during nuclear winter: a preliminary agricultural sector analysis for Aotearoa New Zealand. The New Zealand Medical Journal (online) 136:65–81. https://doi.org/10.1101/2022.05.13.22275065

Wilson RE, Farley SD, McDonough TJ, et al (2015b) A genetic discontinuity in moose (*Alces alces*) in Alaska corresponds with fenced transportation infrastructure. Conservation Genetics 16:791–800. https://doi.org/10.1007/s10592-015-0700-x

Wittmer HU, Ahrens RNM, McLellan BN (2010) Viability of mountain caribou in British Columbia, Canada: Effects of habitat change and population density. Biological Conservation 143:86–93. https://doi.org/10.1016/j.biocon.2009.09.007

Wolfaardt AC, Underhill LG, Altwegg R, et al (2008) African Journal of Marine Science Impact of the Treasure oil spill on African penguins *Spheniscus demersus* at Dassen Island: case study of a rescue operation. African Journal of Marine Science 30:405–419. https://doi.org/10.2989/A

Woltz HW, Gibbs JP, Ducey PK (2008) Road crossing structures for amphibians and reptiles: Informing design through behavioral analysis. Biological Conservation 141:2745–2750. https://doi.org/10.1016/j.biocon.2008.08.010

Wright PGR, Croose E, Macpherson JL (2022) A global review of the conservation threats and status of mustelids. Mammal Review 52:410–424. https://doi.org/10.1111/mam.12288

Xie Z, Yan J (2008) Kernel Density Estimation of traffic accidents in a network space. Computers, Environment and Urban Systems 32:396–406. https://doi.org/10.1016/j.compenvurbsys.2008.05.001

Xu L, Ao C, Liu B, Cai Z (2023) Ecotourism and sustainable development: a scientometric review of global research trends. Environment, Development and Sustainability 25:2977–3003

Yang SZ, Jin HJ, Wei Z, et al (2009) Bioremediation of Oil Spills in Cold Environments: A Review. Pedosphere 19:371–381. https://doi.org/10.1016/S1002-0160(09)60128-4

Yeung P, White B, Ziccardi M, Chilvers BL (2021) What Helps Oiled Wildlife Responders Care for Animals While Minimizing Stress and Compassion Fatigue. Animals 11:1952. https://doi.org/10.3390/ani11071952

York AE (1987) On comparing the population dynamics of fur seals. Status, biology, and ecology of fur seals (JP Croxall and RL Gentry, eds.).

Zeller KA, Wattles DW, Destefano S (2020) Evaluating methods for identifying large mammal road crossing locations: black bears as a case study. Landscape Ecology 35:1799–1808. https://doi.org/10.1007/s10980-020-01057-x

Zhang H, Zhang C, Hu T, et al (2020) Exploration of roadway factors and habitat quality using InVEST. Transportation Research Part D: Transport and Environment 87:102551. https://doi.org/10.1016/j.trd.2020.102551

Zheng B, Liu X, Tang M, et al (2019) Use of age-stage structural models to seek optimal Wolbachia-infected male mosquito releases for mosquito-borne disease control. Journal of Theoretical Biology 472:95–109. https://doi.org/10.1016/j.jtbi.2019.04.010

Ziccardi MH, Wilkin SM, Rowles TK, Johnson S (2015) Pinniped and Cetacean Oil Spill Response Guidelines. NOAA Technical Memorandum NMFS-OPR-52 December 2015.

Zipkin EF, DiRenzo G V. (2022) Biodiversity is decimated by the cascading effects of the amphibian-killing chytrid fungus. PLoS Pathogens 18:e1010624. https://doi.org/10.1371/journal.ppat.1010624

Appendices

Appendix 1: Monthly direct counts at Kaikōura New Zealand fur seal colonies in the 2021/22 and 2022/23 breeding seasons.

Results of monthly direct counts at Ōhau Point colony in the 2021/22 and 2022/23 breeding seasons. * One section of the Ōhau Point colony could not be direct counted in February 2022 due to road works.

Date	December 2021 (10.12.21)	January 2022 (07.01.22)	February 2022 (16.02.22)	December 2022 (12.12.22)	January 2023 (20.1.23)	February 2023 (12.2.23)
Mean live pup counts ± SD	551 ± 16	994 ± 20	716 ± 9*	310 ± 4	902 ± 64	845 ± 14

Results of direct counts at South Rakautara colony in the 2021/22 and 2022/23 breeding seasons. * Only one count was possible in December 2021 as the NZFS stampeded in reaction to counter presence.

Date	December 2021 (16.12.21)	January 2022	February 2022	December 2022 (12.12.22)	January 2023 (17.1.23)	February 2023 (12.2.23)
Mean pup count ± SD	102 *	-	-	50	126 ± 9	72 ± 4

Results of monthly direct counts at the Kaikōura Peninsula subcolonies in the 2021/22 and 2022/23 breeding seasons.

	Date	December 2021 (13.12.21)	January 2022 (11.01.22)	February 2022 (17.02.22)	December 2022 (09.12.22)	January 2023 (19.1.23)	February 2023 (11.2.23)
Mean count ± SD at each subcolony	Lynch’s Reef	2	4	7 ± 4	3	3	1
	KP 2	62 ± 3	58 ± 3	57 ± 2	62 ± 9	100	115 ± 3
	Rhino Horn	34	76 ± 2	75 ± 7	37	81	79 ± 4
	Whalers Bay	154 ± 4	91 ± 4	51 ± 4	43 ± 8	90 ± 4	75

Results of monthly direct counts at the South Coast (SC) subcolonies in the 2021/22 and 2022/23 breeding seasons. * One area of SC1 was not surveyed in December 2021.

	Date	December 2021 (11.12.21)	January 2022 (08.01.22)	February 2022 (18.02.22)	December 2022 (13.12.22)	January 2023 (18.1.23)	February 2023 (11.2.23)
Mean count ± SD at each subcolony	SC1	31 ± 2 *	49 ± 1	61 ± 8	39 ± 1	90 ± 2	79 ± 1
	SC2	160 ± 4	173 ± 3	323 ± 17	110 ± 11	230 ± 3	204 ± 10
	SC3	4	13	15 ± 2	26	29	29
	SC4	34	56 ± 1	66 ± 2	66 ± 3	95 ± 9	80 ± 2
	SC5	1	1	18 ± 2	2	6	13

Results of monthly direct counts at Otumatu in the 2021/22 and 2022/23 breeding seasons.

Date	December 2021 (14.12.21)	January 2022 (08.01.22)	February 2022 (18.02.22)	December 2022 (13.12.22)	January 2023 (18.1.23)	February 2023 (11.2.23)
Mean pup count ± SD	5	0	8	1	3	3

Appendix 2: Choice of variables selected for analysis in the spatial model of NZFS incidents on State Highway 1

Variable	Justification for inclusion in the model
Road width	Wider stretches of road have been linked to increases in wildlife vehicle collisions (WVCs) in other species.
Distance to the nearest bend	Sharp bends in the road may obscure a driver's vision and provide less time to react to NZFS on the road.
Distance to the nearest stream/culvert	NZFS are known to occasionally follow waterways in land, so it was considered that a road segment near a natural water source may have increased probability of experiencing NZFS incidents than one further from water sources.
Road segment height	This was used to demonstrate how high a NZFS would have to have climbed from sea level to reach a given road segment. A measure of slope would have been preferred, but collecting these data points would have involved researchers spending longer on the road and was deemed unsafe.
Distance to beach head	To ascertain whether road segments closer to the coast had a greater probability of experiencing NZFS incidents.
Roadside vegetation coverage (within 5 metres of road)	NZFS have been observed sleeping in bushes near the road, and so it was thought that greater vegetation cover may mean NZFS getting closer to the road, and subsequently onto it.
Location of railway line relative to road	It was thought, if coastal to the road, the railway line would represent another barrier/hazard for NZFS to cross, and so may have reduced the number accessing the road.
Road segment abuts known NZFS breeding area	This was to test whether breeding sites vs. non-breeding haul-outs vs. areas where NZFS are not known to haul at all out influenced incident probability.
Coastal substrate type	To test whether rocky shores, the preferred coastal habitat of NZFS, had a greater probability of NZFS incidents occurrence than sandy beaches.
Non-barriered road	To determine whether the barrier types, or lack thereof, currently in place along the Kaikōura coastline, impacted NZFS incident occurrence.

Single guard rail	To determine whether the barrier types, or lack thereof, currently in place along the Kaikōura coastline, impacted NZFS incident occurrence.
Single guard rail combined with plastic mesh safety fencing (PMSF)	To determine whether the barrier types, or lack thereof, currently in place along the Kaikōura coastline, impacted NZFS incident occurrence.
Motorcycle protection railing (MPR)	To determine whether the barrier types, or lack thereof, currently in place along the Kaikōura coastline, impacted NZFS incident occurrence.
Agricultural fencing	To determine whether the barrier types, or lack thereof, currently in place along the Kaikōura coastline, impacted NZFS incident occurrence.
Standalone PMSF	To determine whether the barrier types, or lack thereof, currently in place along the Kaikōura coastline, impacted NZFS incident occurrence.
Sea wall	To determine whether the barrier types, or lack thereof, currently in place along the Kaikōura coastline, impacted NZFS incident occurrence.

Appendix 3: Media releases on the additional roadside barriers implemented on State Highway 1 following the analyses in Chapter 3

3.1 Report from Chris Lynch Media

Copied from https://www.chrislynchmedia.com/news-items/fur-seal-pups-and-motorcyclists-benefit-by-additional-guardrail-kaikoura/

A stretch of coastline at Kaikōura has become a safer place for both fur seal pups and road users after a joint effort by NZ Transport Agency, Department Of Conservation and Massey University.

A lower band of protective guardrail has been added to existing guardrail at eight sites along the coastline in recent weeks.

The sites were among those identified as being high risk for fur seal pups getting onto SH1 in a study published by Massey University PhD student Alasdair Hall.

The NZTA team, led by Jessica Swift and supported by Carol Bannock, were keen to do what they could to manage the risk to road users and fur seals by safely separating seals from the highway and fast-moving traffic.

By combining DOC's findings on where fur seals were most likely to be hit by vehicles on SH1 and NZTA's ongoing highway maintenance programme, eight stretches of coast were identified where a lower-level guardrail could be retrofitted to existing higher rails. This would stop seals getting onto the road.

The lower-level guardrail is used in many other places on New Zealand highways primarily to protect motorcyclists from sliding under the guardrail in the event of a crash or getting trapped in the guardrail fence. This type of protective railing also increases safety for all road users not just motorcyclists.

The aim was to get the additional guardrail installed in time for the late autumn/winter period when pups are most likely to be killed on the road. Teams from DOC and NZTA worked closely together to confirm site locations, length of guardrail needed (1.2km) then get a fencing contractor and the work underway.

By early May, the additional, lower guardrail had been installed at all eight sites. During a site visit by DOC and the NZTA project team during the installation of the lower-level guardrails, seven dead pups were counted in two small stretches of the road. Had the guardrail been in place a week earlier, they would likely not have made it onto the road and been killed.

3.2 Photo of a report in the Kaikōura Star

Kaikoura Star

Thursday, May 30, 2024

Additional guardrail to protect Kaikoura seals

The new barriers with a double layer to protect fur seals and road users.

PICTURES: Supplied

Fur seals on the Kaikoura coast.

A stretch of coastline at Kaikoura has become a safer place for both fur seal pups and road users after a joint effort by the NZ Transport Agency, Department of Conservation and Massey University.

A lower band of protective guardrail has been added to existing guardrail at eight sites along the coastline in recent weeks.

The sites were among those identified as being high risk for fur seal pups getting onto SH1 in a study published by Massey University PhD student Alasdair Hall.

The NZTA team, led by Jessica Swift and supported by Carol Bannock, were keen to do what they could to manage the risk to road users and fur seals by safely separating seals from the highway and fast-moving traffic.

By combining DOC's findings on where fur seals were most likely to be hit by vehicles, eight stretches of coast were identified where a lower-level guardrail could be retrofitted to existing higher rails. This would stop seals getting onto the road.

The lower level guardrail is used in many other places on New Zealand highways primarily to protect motorcyclists from sliding under the guardrail in the event of a crash, or getting trapped in the guardrail fence. This type of protective railing also increases safety for all road users not just motorcyclists.

The aim was to get the additional guardrail installed in time for the late autumn/winter period when pups are most likely to be killed on the road. Teams from DOC and NZTA worked closely together to confirm site locations, length of guardrail needed (1.2km) then get a fencing contractor and the work under way.

By early May, the additional, lower guardrail had been installed at all eight sites.

During a site visit by DOC and the NZTA project team during the installation of the lower-level guardrails, seven dead pups were counted in two small stretches of the road.

Had the guardrail been in place a week earlier, they would likely not have made it onto the road and been killed.

DOC and NZTA are confident that the fencing will see a notable decrease in the number of fur seal incidents on SH1 in the future, and an added layer of protection for motorcyclists and all road users on this spectacular stretch of SH1.

Ryan Sutherland, who works for NZTA, was involved as an environmental advisor on the Kaikoura Earthquake Recovery project, repairing SH1 and the rail line in the aftermath of the earthquake. This included helping to shift fur seals out of harms way from big, mobile machinery. This year, he was again able to help fur seals.

Joint investigations by Massey University Veterinary Professor Wendi Roe, working with Biosecurity New Zealand investigators and DOC into the deaths of more than 1000 fur seal pups last spring along the Kaikoura coast, found that starvation was why these animals had died.

To learn more, DOC organised an urgent assessment and Mr Sutherland was involved in recording data, capturing, measuring and weighing pups.

"It was certainly exhausting and smelly work, but I feel privileged to have helped, and as everyone who looks into the eyes of a pup would agree, absolutely worth it," he said.

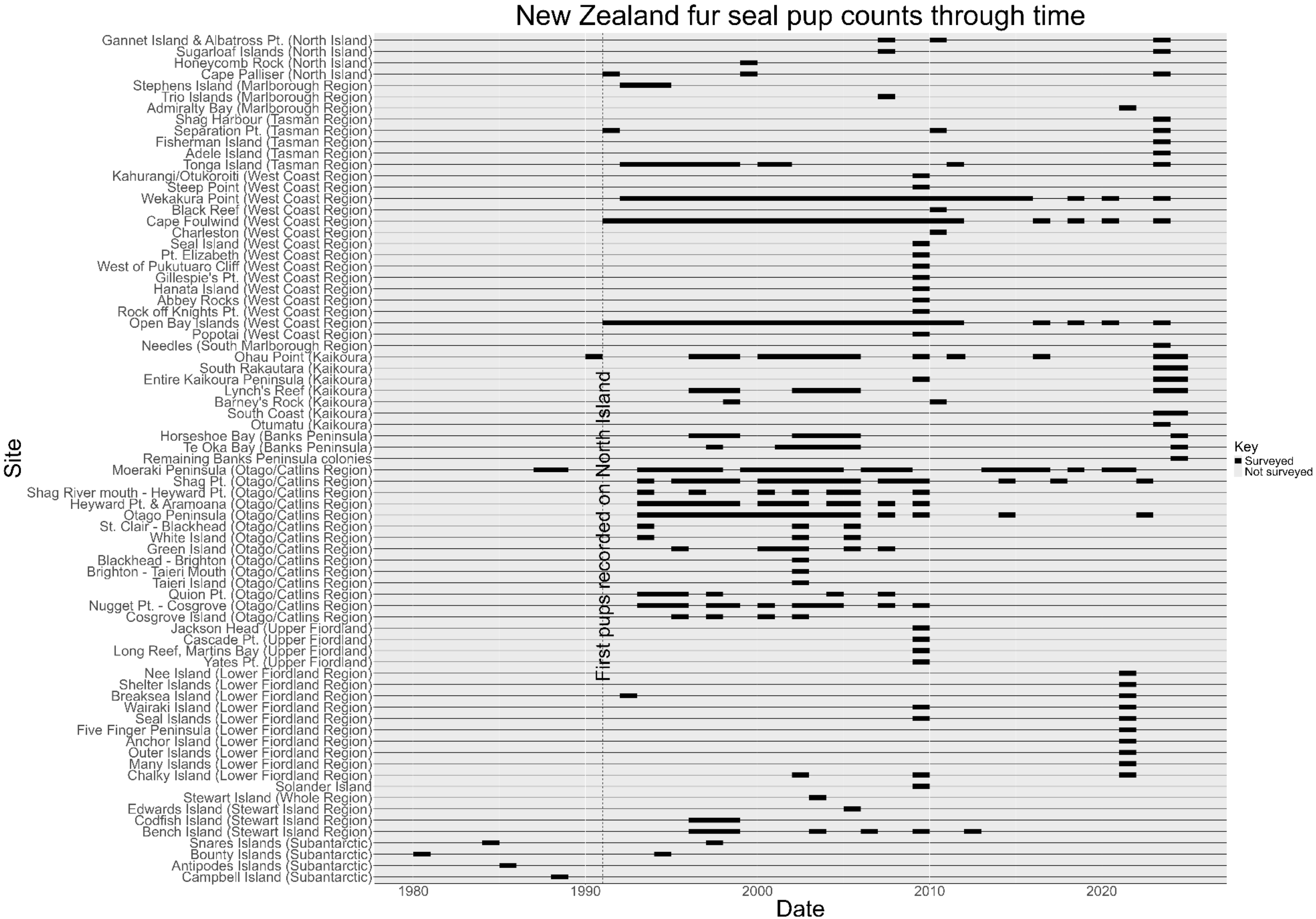
The field team was led by Dr Jody Weir, a marine science advisor for DOC who has worked with Kaikoura wildlife for the past 20 years. With 21 volunteers over three weeks in February this year, she concluded that the Ohau colony had fewer than half the pups produced than in the previous year.

"It's difficult to unravel the causes of starvation, but essentially the fur seals aren't finding enough food to produce and then sustain pups. This is likely due to a combination of factors, including the changing climate and increases in sea surface temperature, not things we can solve quickly," she said.

"On the other hand, we knew that there was something that could be done, with NZTA's help, to significantly reduce the number of fur seal pups killed on this stretch of SH1, by boosting the guardrail protection in these key spots along SH1."

On average 49 seals and pups are hit by vehicles and die on SH1 near the Kaikoura fur seal colonies each year.

Appendix 4: Geographic distribution of New Zealand fur seal abundance studies through time in New Zealand since Wilson (1981).



Appendix 5 Vital rate data used in modelling contemporary New Zealand fur seal abundance in New Zealand.

Study	Species	Cohort (as designated in the study)	Vital rate	Value
(Haase 2004)	New Zealand fur seal	Pups (birth-weaning)	Pup survival	0.73
(Mattlin 1978b)	New Zealand fur seal	Pups (birth – 300 days)	Pup survival	0.6
(Goldsworthy and Page 2007)	New Zealand fur seal	Pups (0-1)	Pup survival	0.557
(Chambellant et al. 2003)	Subantarctic fur seal (*Arctocephalus tropicalis*)	Pups (0-1)	Pup survival	0.667
(Pacoureau et al. 2017)	Subantarctic fur seal	Pups (birth – weaning)	Pup survival	0.58
(Perez-Venegas et al. 2021)	South American fur seal (*Arctocephalus australis*)	Pups (0-1)	Pup survival	0.8
(Lima and Paez 1997)	South American fur seal	Pups (0 – 1.5)	Pup survival	0.59
(Reid and Forcada 2005)	Antarctic fur seal	Pups (0 – 1)	Pup survival	0.776
(Payne 1977)	Antarctic fur seal	Pups (0 – 1)	Pup survival	0.645
(Thomson et al. 2000)	Antarctic fur seal	Pups (0 – 1)	Pup survival	0.71 (1984) 0.86 (1985) 0.78 (1986) 0.71 (1987) 0.75 (1988) 0.79 (1989) 0.81 (1990) 0.68 (1991) 0.93 (1992) 0.79 (1993) 0.53 (1994)
(Goldsworthy and Page 2007)	New Zealand fur seal	Juveniles (1 – 4)	Juvenile survival	0.883
(Beauplet et al. 2005)	Subantarctic fur seal	Post-weaning survival (1 – 3)	Juvenile survival	0.66
(Perez-Venegas et al. 2021)	South American fur seal	Juveniles (1 – 3)	Juvenile survival	0.844
(York 1987)	Northern fur seal (*Callorhinus ursinus*)	Immature females (2 – 4)	Juvenile survival	0.85
(Gibbens and Arnould 2009)	Australian fur seal (*Arctocephalus pusillus doriferus*)	Juveniles (1 – 4)	Juvenile survival	0.86867
(Boyd et al. 1995)	Antarctic fur seal	Individuals (2 – 4)	Juvenile survival	0.88

(Mckenzie 2006)	New Zealand fur seal	Adults females (4+)	Adult survival	0.847 (2001) 0.782 (2002) 0.915 (2003)
(Goldsworthy and Page 2007)	New Zealand fur seal	Adults females (4+)	Adult survival	0.83
(Boyd et al. 1990)	Antarctic fur seal	Adult females (5+)	Adult survival	0.66 (1988) 0.88 (1971-1973)
(Payne 1977)	Antarctic fur seal	Breeding adult females	Adult survival	0.898
(Boyd et al. 1995)	Antarctic fur seal	Breeding adult females	Adult survival	0.652 (1983/84) 0.863 (1984/85) 0.911 (1985/86) 0.731 (1986/87) 0.816 (1987/88) 0.930 (1988/89) 0.752 (1989/90) 0.809 (1990/91 0.862 (1991/92) 0.843 (1992/93)
(York 1987)	Northern fur seal	Adult females (4+)	Adult survival	0.9
(Wickens and York 1997)	Northern fur seal	Females (3+)	Adult survival	0.894
(Frisman et al. 1982)	Northern fur seal	Females (3+)	Adult survival	0.855
(Lima and Paez 1997)	South American fur seal	Females (4.5+)	Adult survival	0.84
(Perez-Venegas et al. 2021)	South American fur seal	Females (4+)	Adult survival	0.83
(Mckenzie et al. 2005)	New Zealand fur seal	Mature females	Pregnancy rate	0.686 (2000) 0.869 (2001) 0.962 (2002)
(Dickie and Dawson 2003)	New Zealand fur seal	Females (4+)	Fecundity	0.69
(Goldsworthy 1994)	New Zealand fur seal	Females	Fecundity	0.67
(Boyd et al. 1995)	Antarctic fur seal	Breeding females	Pregnancy rate	0.661 (1983/84) 0.711 (1984/85) 0.595 (1985/86) 0.623 (1986/87) 0.635 (1987/88) 0.588 (1988/89) 0.798 (1989/1990) 0.622 (1990/91) 0.7650 (1991/92) 0.854 (1992/93) 0.882 (1993/94)
(Frisman et al. 1982; York 1987)	Northern fur seal	Females (3+)	Pregnancy rate	0.72
(Lander 1981)	Northern fur seal	Females (4+)	Pregnancy rate	0.69
(Majluf 1992)	South American fur seal	Unknown	Pregnancy rate	0.82
(Lima and Páez 1995)	South American fur seal	Females (5-15)	Pregnancy rate	0.8
(Perez-Venegas et al. 2021)	South American fur seal	Females (3+)	Unknown	0.622